The Dredge Fishery for Scallops in the United Kingdom (UK): Effects on Marine Ecosystems and Proposals for Future Management

Leigh M. Howarth and Bryce D. Stewart
Environment Department, University of York

Marine Ecosystem Management Report no. 5
May 2014

THE UNIVERSITY of York

***Suggested citation*:** Howarth, L. M. & Stewart, B. D. (2014) The dredge fishery for scallops in the United Kingdom (UK): effects on marine ecosystems and proposals for future management. Report to the Sustainable Inshore Fisheries Trust. Marine Ecosystem Management Report no. 5, University of York, 54 pp.

Front cover: Top image and left images courtesy of Bryce Stewart, right image courtesy of Howard Wood.

Executive Summary

The king scallop fishery is the fastest growing fishery in the UK and currently the second most valuable. The UK is also home to the largest queen scallop fishery out of all of Europe. However, concerns have been raised about the effects of this recent growth of UK scallop fisheries among scientists and conservation bodies, as well as amongst the public following recent media campaigns (e.g. Hugh's Fish Fight). This is because the majority of scallop landings (95%) are made by vessels towing scallop dredges, a type of fishing gear known to cause substantial environmental impacts. In addition, several scallop stocks are showing signs of overexploitation and there is concern over future impacts of ocean warming and acidification. Although, there have been several recent improvements in the management of scallop fisheries in parts of the UK, information on many scallop stocks around the UK is still lacking. This report therefore proposes that better monitoring and stock assessments are needed for these scallop fisheries and stocks. With recent legislation soon to result in the development of a new network of marine protected areas (MPAs) around the UK, and improved management of fisheries in European Marine Sites, now is a crucial time to review the UK scallop dredge fishery and its impacts on the wider environment so that this new legislation can support a sustainable future for the UK scallop fishery. This report was therefore commissioned by the Sustainable Inshore Fisheries Trust with the aim of collating existing knowledge on the management and environmental impacts of scallop fisheries around the UK.

Of all the fishing gears, scallop dredges are considered to be the most damaging to non-target benthic communities and seafloor habitats. This report documents that the impacts of scallop dredging can vary greatly between different seabed types. Slow-growing organisms that form biogenic reefs, such as maerl and horse mussels, are the most vulnerable to scallop dredging and their damage can have severe consequences on local and regional biodiversity. Therefore, there is a strong argument for completely protecting biogenic reefs from all towed fishing gear. Ecological communities on soft sediments can also be impacted by scallop dredging. However, communities located in high energy environments are generally more resistant to natural disturbance and towed fishing gears. Rocky reefs, although generally avoided by scallop dredgers, can also suffer damage from scallop dredging. However damage tends to incremental, increasing with the number of dredge tows performed.

The impacts of dredging also vary between different groups of organisms. The benthic epifauna (i.e. the organisms that attach to the seabed) are most vulnerable to scallop dredging and their removal / damage can greatly reduce an area's capacity to support biodiversity and can negatively impact upon the recruitment of commercially important species, including scallops themselves. Mobile species can also be affected by dredging. Dredges can create considerable levels of by-catch in a large number of commercial and non-commercially targeted species, the majority of which is discarded damaged, dying or dead. They also cause considerable levels of damage and mortality to those organisms impacted by the dredge but left uncaught on the seabed. These patterns can therefore give rise to considerable conflict between scallop fisheries and fisheries targeting other species. In contrast, studies report inconsistent results of dredging on burrowing infaunal communities, with some observing no effect, and others reporting strong changes to infaunal abundance and biodiversity. The use of scallop dredges can also cause considerable physical impacts to the seabed, such as homogenization and resuspension of sediments, and cause alterations in seabed topography and nutrient cycling.

Considering the conflicts and considerable environmental impacts associated with scallop dredging, there is an urgent need for better management of scallop fisheries in the UK. This report presents a number of case studies of successful management of scallop fisheries in the UK that resolve gear conflict and their ecosystem impacts. These case studies include:

- The South Devon Inshore Potting Agreement (IPA) in England. The South Devon IPA separates static and mobile fishing gears from large areas of the seabed. Despite initial concerns, the management system has remained stable for over 35 years. The system is widely regarded as a success by both fishers and managers because it has effectively allowed fishers from both sectors to operate profitably on traditional fishing grounds. An unplanned for, but welcome side effect of this agreement has been considerable benefits to marine biodiversity in the areas where towed gears have been excluded. Responses have included significant increases in the biomass of hydroids, soft corals and other important nursery habitats, as well as increases in long-lived molluscs and large burrowing urchins. Scallop densities have also increased within the areas closed to towed gears, potentially increasing scallop recruitment both inside and outside the protected areas, as well as a number of fish species which have also increased in abundance.

- The Port Erin Closed Area, Isle of Man. A small 2 km^2 area was closed to towed gears (and taking of scallops by any means) off the Isle of Man to monitor the response of the benthic community in the absence of fishing. After seventeen years of protection, king scallop densities were thirty times greater within the closed area than when first protected. The reduction in fishing mortality also allowed individuals within the closed area to reach much older and larger sizes, with exploitable and reproductive biomass of scallops being 20 and 33 times higher respectively, than on the adjacent fishing ground. There is also growing evidence that export of larval scallops from high rates of breeding within this closed area has boosted surrounding populations and therefore the fishery. Overall, scallop catch rates have reached a 20 year high on many fishing grounds despite the local fleet being half the size it was in the early 1980s. Not only does the closed area appear to have helped king scallop populations recover, it has also led to the development of more diverse and structurally complex epibenthic communities, particularly in terms of upright hydroids and bryozoans. Due to the success of the Port Erin closed area, both in terms of fisheries and conservation benefits, the Isle of Man government has subsequently established a network of similar protected areas around the island. Importantly, the local fishing industry is now strongly supportive of these spatial management measures and is actively involved in related research and monitoring.

- Lyme Bay Marine Protected Area (MPA) in Dorset and Devon, England. Concerns over the impacts of towed fishing gears on rocky reefs in Lyme Bay resulted in the establishment of a large statutory MPA which excluded towed gears from the area. Boulders and cobbles within the newly protected area had limited life growing on them when monitoring first began. However, observations made three years later revealed structural complexity had substantially increased within the MPA through the recovery of pink sea fans (increase of 636%), ross coral (increase of 385%), branched sponges (increase of 414%) and hydroids (increase of 229%). Such species are known to improve survivorship of juvenile fish by acting as important fishery nursery areas and feeding grounds. In addition, the main target species of the excluded fishery, the commercially valuable king scallop, was also found to be in a state of recovery within the MPA.

- Lamlash Bay No-Take Zone (NTZ), Isle of Arran, Scotland. Lamlash Bay is the first and only fully protected NTZ in Scotland, and the only marine reserve in the UK originally proposed for both conservation and fishery objectives. After four years of protection, important nursery habitats were twice as abundant within the NTZ compared to neighbouring fishing grounds, and their abundance has been steadily increasing. The recovery of these habitats was found to result in higher levels of settlement by juvenile scallops meaning juvenile scallop abundance was more than 350% higher within the NTZ than outside in some years. The density of adult king scallops has also increased and evidence suggests that the NTZ is enabling the age and size structure of scallop populations within its boundaries to return to a more natural and extended state as, after four years of protection, it was found that king scallops were on average 25mm larger and 1.6 years older within the NTZ than outside. In further support of this, the reproductive biomass of king scallops was 185% greater within the NTZ than on surrounding fishing grounds. The NTZ also appears to be generating fishery benefits for other commercially important species. Catch rates of legal-sized European lobsters were 189% higher within the NTZ than neighbouring fishing grounds. Furthermore, catch rates, and the weight and size of lobsters were all found to be greater within the reserve and all declined with increasing distance from the boundaries of the reserve, possibly indicating spillover. Later tagging studies confirmed this. In addition, the potential number of eggs carried per female lobster was 27.3% higher within the NTZ and berried (egg-bearing) females were 5.5 times more abundant, suggesting that the 2.67 km^2 NTZ has a potential egg output equivalent to an unprotected area of 19.1 km^2.

By damaging seafloor habitats, scallop dredging not only significantly reduces biodiversity; it also damages much of the habitat that is crucial for the settlement and survival of juvenile scallops, as well as a number of other species of commercial importance. We therefore conclude that there is considerable evidence that the management of UK scallop fisheries could be significantly improved. A new management regime for UK scallop fisheries that provided better protection to vital scallop nursery and breeding areas would undoubtedly result in more productive and sustainable fisheries, and maintain healthier benthic ecosystems.

Excluding scallop dredging from selected areas of the seabed can resolve conflict between fisheries and generate ecological and fishery benefits. In the case studies presented, the benefits of excluding towed fishing gears have outweighed the costs of losing access to some fishing grounds. We therefore believe that a network of protected areas around the UK, both including and beyond what is currently in place and being proposed, would provide substantial benefits to the scallop fishery, and reduce its impact on the wider ecosystem. The following principles should be used to guide the development of this network:

- Protected areas should be strategically located and designed to offer multiple benefits wherever possible.

- Scallop dredging should be excluded from vulnerable habitats within existing and future protected areas at the site level, rather than just specifically where vulnerable features currently exist.
- Protected areas should not just cover the most vulnerable habitat types, but ensure representation of the full of range of substrates and biodiversity.
- Protected areas should be permanent to maximise benefits to fisheries and conservation.
- Protected areas should be well monitored in order to assess performance.

Closing some areas to fishing may have some short term negative effects on local economies and the welfare of coastal communities. If these short term costs can be overcome, the scallop fishing industry is one of the economic groups with the most to gain in the long term. However, the same industry also has the most potential to impact on the success of this approach. Fishers must therefore be actively involved in the decision making process when closed areas are being established and emphasis should be placed on the fishery benefits that closed areas can afford. Spatial management of the UK scallop dredge fishery, as described above, will go a long way towards ensuring it has a sustainable and productive future while reducing its impact on the wider ecosystem. We also promote reduced fishing efforts overall, development of more environmentally friendly dredges and local management of scallop fisheries, particularly in the inshore sector, to encourage enhanced levels of stewardship within the industry.

Table of contents:

1. Scallop Fisheries around the United Kingdom

1.1. Biology

Two species of scallop (Pectinidae) support important commercial fisheries in the United Kingdom (UK); the larger and more valuable king or great scallop, *Pecten maximus,* and the smaller queen scallop, *Aequipecten opercularis* (Fig. 1). The biology of the two species is quite different and this influences both the productivity of their fisheries and methods of exploitation.

Figure 1 | The king or great scallop, *Pecten maximus* (left - *photo: Bryce Stewart*) and queen scallop, *Aequipecten opercularis* (right - *photo: Howard Wood*).

Both species begin their life with the release of gametes during spring / summer spawning events (Brand 2006a), followed by fertilisation, embryonic and larval stages (LePennec et al. 2003). The resulting free-swimming larvae typically spend 3–6 weeks in the water column, often dispersing over considerable distances (Brand et al. 1980; Macleod et al. 1985) before eventually settling on to the seabed. There they attach to the seafloor and undergo their final transition into the free-swimming adult form (Brand 2006a). As a result, the reproductive success and recruitment of scallops (i.e. the number of individuals surviving juvenile development and entering the fishery) is influenced by a multitude of factors including spawning stock biomass, the availability of suitable settlement habitat, environmental conditions, and ecological interactions such as predator density (Beukers-Stewart et al. 2003; LePennec et al. 2003; Brand 2006b; Beukers-Stewart & Beukers-Stewart 2009). Once they reach adulthood, king scallops are relatively static, rarely moving more than 30 m in 18 months and characterised by predictable patterns of distribution (Howell & Fraser 1984). In contrast, queen scallops are known to be much more mobile than king scallops (Jenkins et al. 2003; Brand 2006b), although this has not been fully quantified.

In UK waters, king scallops become sexually mature at approximately 2-3 years old and 80-90 mm in shell length, but may live for over 20 years and grow to over 200 mm in undisturbed populations (Tang 1941). In comparison, queen scallops mature between 1-2 years old and approximately 40 mm in shell length, and rarely live for more than 5-6 years or grow to more than 90 mm (Vause et al. 2006). In general, the longer life span of king scallops leads to more stable population dynamics and predictable distributions than observed for queen scallops. For example, the fishing grounds for king scallops around the Isle of Man have remained remarkably consistent throughout the history of the fishery, with some grounds producing commercially viable quantities every year for over 80 years

(Brand 2006a). In contrast, the productivity of queen scallop fisheries tends to be highly variable both temporally and spatially, with queen scallops sometimes being absent from previously productive fishing grounds for more than a decade before appearing again in high densities (Vause et al. 2007).

1.2. Significance and distribution

In the UK, landings of the king scallop (*Pecten maximus*) are growing faster than any other commercially targeted species. King scallop landings increased from 14 thousand tonnes in 1994, to 53.3 thousand tonnes in 2012, a rise of 281% (Fig. 2). During the same period, the first sale value of king scallops rose by 216%, increasing from £21.2 million in 1994 to £66.9 million in 2012 (Radford 2013). Due to this recent rise in landings and value, king scallops are now the UK's second most valuable fishery resource.

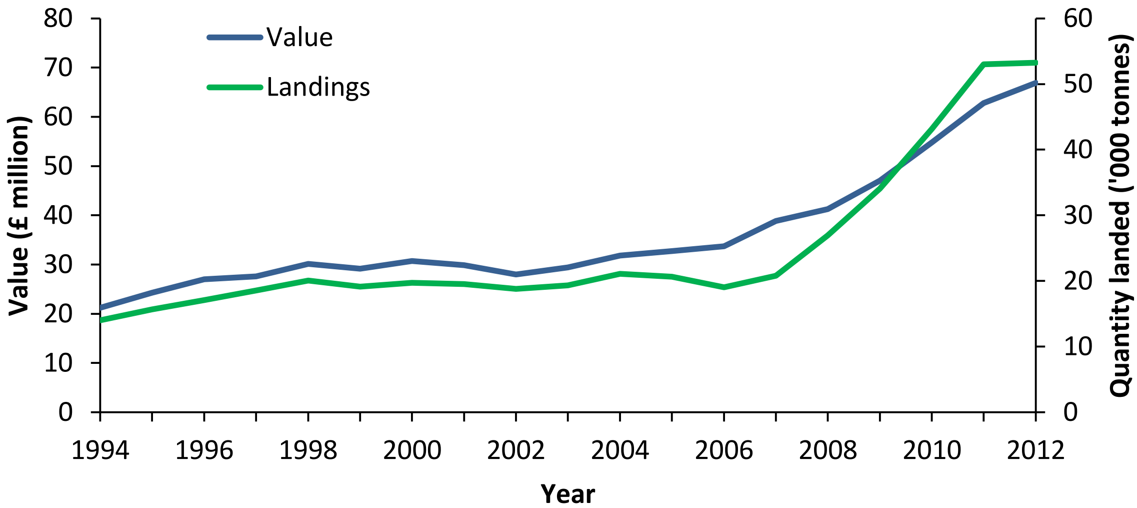

Figure 2 | The value and quantity of king scallops landed by UK vessels between 1994 and 2012. Data obtained from the Marine Management Organisation (MMO).

Queen scallops (*Aequipecten opercularis*) are also commercially targeted in some parts of the UK. In general, landings of queen scallops are more variable and much less valuable than king scallops, with 12 thousand tonnes, worth £4.6 million pounds, landed in 2010 (MMO 2012). The years between 2010 and 2012 saw unusually high catches of queen scallops in the UK, with over 10 thousand tonnes captured each year in the Irish Sea alone, however, the fishery now appears to be in decline (Murray 2013). Nonetheless, even with landings averaging at just 6 thousand tonnes a year over the last decade, the UK is responsible for landing the largest quantity of queen scallops in Europe (Beukers-Stewart & Beukers-Stewart 2009).

King and queen scallop fisheries are predominantly distributed around the western parts of the UK (Brand 2006a; Capell et al. 2013). Very few king scallops are taken in the mid or southern North Sea (Beukers-Stewart & Beukers-Stewart 2009); although a moderate seasonal fishery has developed off the Yorkshire coast over the last decade (Brand 2006a; Capell et al. 2013). Instead, the main fisheries for king scallops are concentrated in the eastern and western English Channel, the Irish Sea, and off the west and north-east coasts of Scotland (Brand 2006a). Scallop stocks located around Scotland account for over half of the UK king scallop fishery, and consequently, Scottish boats are responsible for approximately half of the UK catch (Dobby et al. 2012; Capell et al. 2013). In comparison, queen scallop fisheries are mostly concentrated in the Irish Sea and off the west coast of Scotland. Relatively few boats target queen scallops in the English Channel or the North Sea (Brand 2006a).

1.3. Scallop fleet

In 2012, the UK fishing fleet comprised of 6,406 vessels. Of these, 4,246 vessels (66% of the fleet) were less than 10 m in length, of which only two had licenses to fish for scallops. However, out of the remaining 1,273 vessels greater than 10 m in length, 381 vessels (30% of the over 10 m fleet) were licensed to fish for scallops. Despite the over 10 m fleet accounting for only 30% of the UK fishing fleet, they are responsible for 52% of all fishing effort in UK waters (Radford 2013).

Vessels towing scallop dredges were one of the few fishing gears unaffected by decommissioning exercises carried out by UK fisheries administrations between 2001 and 2003. Consequently, fishing effort by scallop vessels increased by 18% between 2002 and 2012 (Fig. 3). In contrast, fishing effort by beam and otter trawls declined by 42% during this period (Radford 2013). Overall, the recent rise in scallop landings and fishing effort are thought to have been driven by a combination of favourable stock levels, tight quotas on most of the main alternative species (i.e. finfish), the consistently high market value of king scallops, and the predominately inshore distribution of scallops which results in lower fuel costs than those associated with offshore fisheries (Beukers-Stewart & Beukers-Stewart 2009).

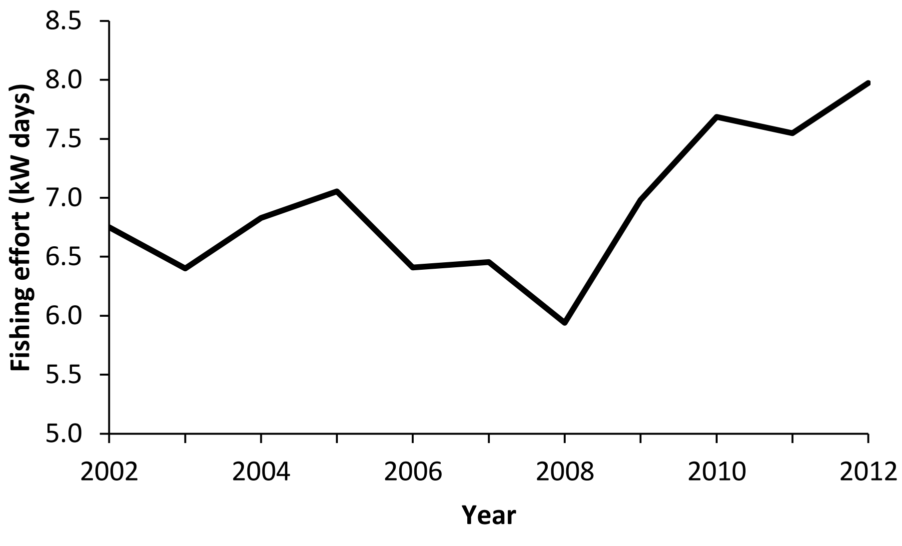

Figure 3 | Fishing effort (kW days) of the UK scallop dredging fishing fleet between the years 2002 to 2012. Data obtained from the Marine Management Organisation (MMO).

1.4. Status of stocks

At face value, it may appear that UK scallop stocks are healthy because landings are continuing to rise with increasing fishing effort. There is even some evidence that current rates of ocean warming are increasing king scallop recruitment due to the positive effects warming can have on gonad and larval development (Shephard et al. 2010). However, in parts of the UK where stock assessments have been conducted, the picture is somewhat patchy. The king scallop fishery in Shetland, Scotland, was certified as sustainably managed by the Marine Stewardship Council (www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/shetland-inshore-crab-lobster-and-scallop) in 2012. In contrast, concerns have recently been raised over increasing mortality, declining recruitment and spawning stock biomass in several other major Scottish stocks (Hall-Spencer & Moore 2000; Howell et al. 2006; Hinz et al. 2011; Barreto & Bailey 2013). Likewise, although king scallop stocks are not routinely monitored or assessed in England, Wales or Northern Ireland at present, some stocks are thought to be showing signs of decline (MMO 2012). Very little is known about the state of queen scallop stocks in the UK, apart from around the Isle of Man where regular surveys are conducted and the trawl fishery was certified as sustainable by the Marine Stewardship Council (MSC) in 2011 (www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/Isle-of-Man-queen-scallop). However, despite this endorsement the fishery now shows signs of over-exploitation (Murray 2013) and the certification was suspended in May 2014 (see below).

1.5. Methods of exploitation

Newhaven dredges:

There are three main methods of harvesting wild scallops in the UK. By far the most widely used are scallop dredges. Over 95% of all king scallops landed in the UK are caught by "Newhaven" scallop dredges (Barreto & Bailey 2013; Radford 2013). This method involves towing sets of dredges along the seabed, located either side of the fishing vessel, at speeds of 3-4 knots (Fig. 4a). As king scallops normally live buried within the sediment (Bradshaw et al. 2001), the opening of the dredge is fitted with a spring-loaded bar of 8-9 teeth, each up to 11 cm long and spaced 8cm apart, which are designed to rake scallops out from the sediment and into a dredge net which trails closely behind (Fig. 4b). The teeth on Newhaven dredges penetrate anywhere between 3-10 cm into the seabed depending on seabed type (Kaiser et al. 1996). The spring-loaded tooth bar allows the teeth of the dredge to flex backwards, preventing it from snagging on harder ground and improving catch efficiency. The tension in the springs can also be adjusted to improve the efficiency of the gear on different seabed types (Kaiser et al. 1996). Still, the capture efficiency of Newhaven scallop dredges is quite low – between 5-41 % for legal sized scallops depending on seabed type and operating conditions (Dare et al. 1993; Beukers-Stewart et al. 2001; Jenkins et al. 2001). In the UK, each dredge is normally 75 cm in width and the mesh size of the underside "belly" and top nets are generally 80 mm and 100 mm respectively. Typically the belly of the dredge net is constructed of steel rings in order to reduce damage from rough ground (Kaiser et al. 1996). Vessels can tow anywhere between 2 and 22 dredges per side depending on local regulations and vessel power (see section 1.6). Dredges are typically towed in gangs suspended from a towing bar fitted with rubber wheels designed to roll along the seabed. Thanks to their penetrative nature and close contact with the seabed, the use of Newhaven dredges can cause substantial physical disruption to the seafloor and associated ecological communities (see section 2). Furthermore, as Newhaven dredges are relatively inefficient in capturing targeted king scallops, fishers tend to perform repeated tows within the same area, thereby exacerbating any impacts they have on marine ecosystems (Dare et al. 1993; Beukers-Stewart et al. 2001; Jenkins et al. 2001).

Figure 4 | a) A gang of four spring toothed Newhaven dredges showing the tooth bars, dredge frame and nets, and the rubber wheeled towing bar (*photo: Bryce Stewart*) b) Close up of the dredge teeth.

Some efforts have been made to develop less environmentally damaging dredges, most notably the "Hydrodredge" which is fitted with cups designed to displace scallops with turbulent water flow instead of using teeth (Shephard et al. 2009). Although this dredge reduced levels of damage to scallops and bycatch, the overall catch rates of scallops were only 10 to 40% of those in standard dredges (Shephard et al. 2009). This would make commercial uptake of the Hydrodredge unlikely in its current format.

Diver caught:

On a much smaller scale, around 2 to 4% of king scallops landed in the UK are collected by hand by SCUBA divers (Howell et al. 2006; Dobby et al. 2012). Scallop divers are limited by depth (< 30 m) and air consumption / decompression limits, meaning they fish much smaller areas of the seabed compared to boats towing scallop dredges. Due to the limited numbers of scallops that can be collected during each dive, dive fisheries tend to concentrate on the largest, most valuable individuals to ensure economic viability. Concerns have therefore been raised about the potential impact of dive fisheries on scallop stocks (Kaiser 2007a), as large-bodied scallops contribute disproportionally to recruitment by producing considerably greater quantities of eggs than small scallops (Bradshaw et al. 2001; Beukers-Stewart et al. 2005). However, these concerns should be weighed up against the much larger impacts and quantities captured associated with dredge fisheries.

Skid dredges:

In contrast to king scallops, queen scallops tend sit on the surface of the seabed and are much more mobile, capable of swimming 2-10 metres to avoid disturbance or predators (Brand 2006b; Howarth pers.obs). The fishing gears used to catch queen scallops are therefore slightly different from those used by the king scallop fishery. "Skid dredges" operate in much the same way as Newhaven dredges, but the tooth bar is replaced with a "tickler chain" which disturb queen scallops resting on the seafloor, causing them to swim upwards into the water column where they can be caught by the net (Fig. 5). Also, instead of rubber wheels on the tow bars, skid dredges are fitted with skis or skids designed to run along the top of the seabed.

Figure 5 | A gang of skid or ski dredges showing the four skids on the underside of each dredge, but the absence of a toothed bar (*photo: Harriet Salomonsen*)

Otter trawls:

In some parts of the UK, otter trawls are also used to catch queen scallops. These involve towing a net across the seabed, held open by two trawl doors running in front of the net (Fig. 6). Similar to skid dredges, tickler chains are located on the bottom of the net to disturb resting queen scallops (Løkkeborg 2005). The choice of skid dredges or otter trawls is largely governed by the nature of the substrate on different fishing grounds, with skid dredges being more effective in rough / coarse sediment areas and trawls in sandy / muddy areas (Vause et al. 2007). Either way, both gears take advantage of the natural propensity of queen scallops to swim up into the water column when disturbed, rather than relying on extraction of the scallops from the sediment as is the case for Newhaven dredges. Consequently, the general consensus is that fishing for queen scallops causes less disturbance to the seabed than dredging for king scallops (Collie et al. 2000; Hinz et al. 2012).

Figure 6 | A demersal otter trawl being retrieved with a catch of queen scallops (*photo: Simon Park*)

Around the Isle of Man, where the main UK fishery for queen scallops exists, otter trawlers made 20 to 24% of queen scallop landings in 2010 and 2011, while UK dredgers took over 65% of the catch (Murray 2013). By comparison, in 2012 otter trawlers caught around 63% of total landings within the Manx territorial sea, with dredgers taking 68% of landings across ICES rectangles 36E5 and 37E5 (see Fig. 7; Murray 2013).

1.6. Management

In the UK, there are no limits on scallop landings in the form of Total Allowable Catch (TACs) or quotas. Instead, UK scallop fisheries are controlled predominantly through the use of minimum legal landing sizes, gear restrictions, seasonal closures and some effort controls on the largest boats (> 15 m in length).

For king scallops, current European Union (EU) legislation specifies a minimum landing size of 100 mm shell length, except in the English Channel and Irish Sea where the limit is 110 mm (Barreto & Bailey 2013). Generally, the maximum number of dredges is restricted to 6-8 per side within 6 nautical miles of the shore around the UK (Howell et al. 2006; Cappell et al. 2013). Between 6 and 12 miles the fishery is less restricted, with a maximum of 8 dredges per side allowed in English waters, and 10 per side in Scottish waters (Cappell 2013). Outside the 12 mile limit, up to 14 dredges are permitted in Scotland (Dobby et al. 2012), but in England there are no limits (Cappell et al. 2013). As a result, some boats fish more than 20 per side, with dredge number only being limited by the size and horsepower of the fishing vessels. Scallop fisheries in Wales are more strictly regulated than anywhere else in the UK. No scallop fishing is allowed within 1 mile of the shore and dredging between 1 and 3 miles is only permitted by boats less than 10 m in length and towing no more than 6 dredges in total (Cappell 2013). Within 3-6 miles and 6-12 miles respectively, totals of 8 and 14 dredges are allowed (Cappell 2013). Furthermore, all scallop dredgers in Wales must carry and use working satellite Vessel Monitoring Systems (VMS; Cappell 2013). Throughout Northern Irish waters out to 12 miles there is a maximum limit of 6 dredges per side. Finally, within Manx (Isle of Man) waters, dredges are limited to 25 feet total width out to 3 miles, and 40 feet between 3 and 12 miles. VMS is also compulsory around the Isle of Man (Cappell et al. 2013), which both here and around Wales has been key to developing a better understanding of stock and fleet dynamics and ensuring fisheries regulations are abided by.

Apart from these general regulations, various local authorities impose stricter rules in specific areas throughout the UK. For example, some Inshore Fisheries and Conservation Authorities (IFCAs) in England limit vessel sizes within their jurisdictions to 12-15 m in length. In addition, scallop dredging is banned within 3 miles of the shore in the Sussex IFCA district, and in an increasing number of European Marine Sites - Special Areas of Conservation (SACs) - in both England and Wales (see below). For example, recent byelaws introduced by the Southern IFCA in England, who manage a total area of 670km^2, ban the use of towed fishing gear (including scallop dredges) within 25% of their coastal waters (www.southern-ifca.gov.uk). Likewise, dredging is banned within the Cardigan Bay SAC in Wales (www.cardiganbaysac.org.uk) and in 6 fishery exclusion zones around the Isle of Man (see section 3.2.1). A network of Marine Conservation Zones (MCZs) is currently being introduced in England and Wales through the UK Marine and Coastal Access Act (Scot Gov 2014a), and a network of Scottish Marine Protected Areas (MPAs) is currently being consulted on in Scotland through the Marine (Scotland) Act (Scot Gov 2014b). However, it is still unclear how these designations will regulate fisheries.

These differing restrictions on scallop fishing activity in relation to distance from shore have effectively split the UK scallop fishing fleet into two components. Smaller vessels (8-15 m in length) fishing fewer dredges tend to dominate the inshore sector (within 6 miles of shore) and generally land their catch locally on a daily basis. In comparison, the offshore fleet of large vessels (greater than 15 m in length) operate large numbers of dredges and may fish around the clock for 4 to 5 days at a time. This fleet is often highly nomadic, with some boats fishing right around the UK coastline in response to changing stock availability and regulations (Palmer 2006).

The dredges themselves must also adhere to certain specifications regarding internal belly ring diameter (≥ 72 mm), top net mesh size (≥ 100 mm), tooth number (< 10) and tooth spacing (≥ 75 mm). Additionally, the use of “French” dredges (a design incorporating water deflecting plates and rigid fixed teeth) is prohibited in Scottish inshore waters and ICES areas VII d, e, f and h (see Fig. 7; DEFRA 2013).

Seasonal closures of the king scallop fishery are enforced in the Irish Sea and within 6 miles of the Sussex, Devon, Yorkshire and Welsh coastlines. These closed seasons normally run from July to September, although there are some small variations, and have the effect of protecting scallops during the period when they are breeding and free-swimming larvae are settling on to the seabed. In some areas of the UK, scallop fishing effort is also limited either to certain times of the day (e.g. from 5 am to 9 pm in Northern Ireland and from 6am to 9 pm in Shetland) or through the use of weekend bans (e.g. in Northern Ireland and the Firth of Clyde).

Several measures are also in place that attempt to limit scallop fishing effort. The Western Waters effort regime applies to all UK fishing vessels over 15 metres in length fishing in waters to the west of Scotland, Wales, England (including the English Channel) and south west towards France and Spain (ICES Areas VI, VII and VIII, see Fig. 7; Dobby et al. 2012). Under this regime, the limits for UK vessels were 1,974,425 KW days for ICES Sub-areas V and VI and 3,315,619 KW days for Sub-area VII in 2012 (Dobby et al. 2012). However, when the UK exceeded its effort allocation in the latter area in 2010, 2011 and 2012, extra effort was gained via swaps with other member states (www.marinemanagement.org.uk/fisheries/management/days_western_swaps.htm; Cappell et al. 2013). It is therefore difficult to know if this scheme is actually having the desired practical effect of limiting overall scallop fishing effort each year.

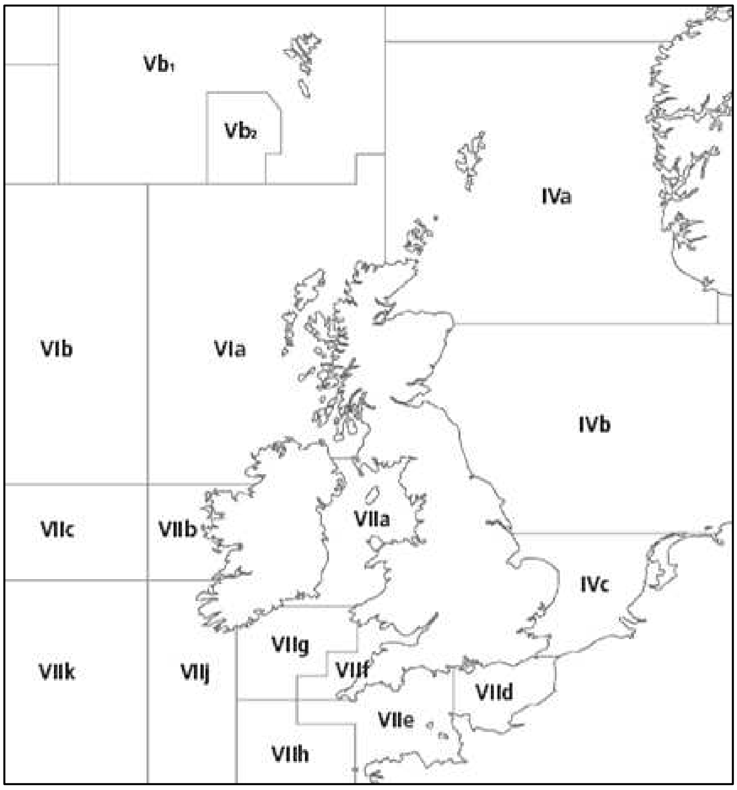

Figure 7 | ICES statistical areas and sub-areas and their position around the UK. Taken from http://geo.ices.dk/

The number of licenses permitting the commercial capture of scallops in the UK was capped for vessels greater than 10 m in 1999. However, there are similar concerns that this has had little effect in limiting fishing effort as far more licences were granted than there were boats participating in the fishery (Brand 2006a). A recent review of the Scottish scallop fishery reported a near unanimous view from stakeholders that effort in the fishery had expanded to unsustainable levels (Cappell et al. 2013). The Scottish government has recently recognised the issue with latent effort and has suggested removing scallop fishing entitlements from boats which have not used them in the past 7 years (Scottish Government 2013). How effective this will be remains to be seen, as boats will have 6 months warning to re-activate their entitlements. However, to do this they will need to fully rig their boats for scallop fishing, a considerable financial investment (Cappell et al. 2013).

In most areas there are few regulations on the UK fishery for queen scallops. The EU minimum size is 40 mm shell height; however, it is generally uneconomic to process queen scallops less than 55 mm. There are no closed seasons for queen scallops or restrictions on fishing time or catches, apart from around the Isle of Man where a range of regulations have been introduced in recent years (Sea Fisheries (Queen Scallop Fishing) Bye-laws 2013) and there are proposals to introduce more www.gov.im/lib/docs/daff/Consultations/2014qscconsultationfinal.pdf). These measures were designed to protect the stock within the Territorial Sea and allow a sustainable fishery in light of dramatic recent increases in effort in the fishery. The measures included an increased minimum landing size of 55mm, an increase in cod end mesh size of trawls to 85mm, the introduction of a weekend ban on fishing and a curfew on fishing from 1800 hours to 0600 hours. Based on stock surveys by Bangor University, a TAC of 4,000 tonnes for the trawl fishery and a further 1,000 tonnes for the dredge fishery was set for 2013. The trawl fishery opened on the 17th of June 2013 and the dredge fishery on 1 October 2013. The trawl fishery was closed at beginning of October and the dredge fishery at the end of November when the TAC was reached (DEFA 2013). As a result of these types of relatively stringent management measures, the Isle of Man queen scallop trawl fishery was certified as sustainable by the Marine Stewardship Council (MSC) in 2011 (www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/Isle-of-Man-queen-scallop).
Unfortunately, despite all of these efforts the latest stock assessment has revealed a dramatic drop in the biomass of queen scallops around the Isle of Man. As a result the MSC certification has now (May 2014) been suspended until further notice http://www.msc.org/track-a-fishery/fisheries-in-the-program/certified/north-east-atlantic/Isle-of-Man-queen-scallop/assessment-downloads-1/20140520_Suspension_Notice_ANMT_SCA61.pdf This situation highlights the difficulty of managing fisheries for species like queen scallops, which undergo considerable natural fluctuations in abundance and have been highly targeted by the UK fishing industry in recent years (Vause et al. 2007; Murray 2013).

Compared to the dredge and trawl fisheries, the UK dive fishery for scallops is subject to very few regulations other than minimum sizes (as above) and the closed seasons for king scallops which apply to both dredgers and divers. Divers fishing for scallops are also excluded from some closed areas (see below and section 3).

By at least partially limiting the intensity of fishing effort, the restrictions mentioned above may indirectly confer some benefits for conservation of the wider environment. However, the current management of UK scallop fisheries generally consists of measures designed to promote the sustainability of scallop stocks, rather than sustaining ecosystems as a whole. The few direct conservation measures currently in place in UK waters predominately take the form of spatial restrictions on the use of towed fishing gears (see above and section 3). Examples include Special

Areas of Conservation (SACs – EU Habitats Directive 1994) in the Fal and Helford (England), Lyn Peninsula and Cardigan Bay (Wales), and the Firth of Lorn (Scotland); Sites of Community Importance (SCI – Fishing Restrictions Order 2008) in Lyme Bay (England); No-Take Zones (NTZs) off Lundy Island, Flamborough Head (England), and Lamlash Bay (Scotland); Marine Nature Reserves (MNRs) off Skomer Island (Wales) and Strangford Lough (Northern Ireland); and the Inshore Potting Agreement (IPA) in south Devon (England). The Department of Environment, Food and Rural Affairs (DEFRA) recently developed a matrix indicating the impacts of different fishing gear on different habitats to guide fisheries management policy in European Marine Sites (EMS) such as Special Areas of Protection (SPAs) and SACs (MMO 2014). The matrix identified scallop dredging to be damaging to seagrass beds, maerl beds, chalk reefs, boulder reefs, *Saballaria* reefs and a number of other habitats, meaning scallop dredging may become completely banned from such habitats (DEFRA 2013b; Cappell et al. 2013). This has already happened within the jurisdictions of several IFCAs (see above) however, a contentious and unresolved issue is whether to only ban dredging on the vulnerable features (i.e. specific habitats) within EMS (and other MPAs as they develop – see above), or from the sites completely. Experience from Lyme Bay (see section 3.2.2; Sheehan et al. 2013a) and ecological theory (Rees et al. 2013) makes a strong case for excluding damaging activities such as scallop dredging at the site level. However, this is likely to meet with resistance from the fishing industry as it will mean a greater loss of potential fishing grounds.

1.7. Gear conflict

In the UK, many different species of fish and shellfish are often targeted within the same areas (MMO 2012). However, when two or more species coexist in a marine habitat, conflict may arise between different sectors of the fishing industry. This is particularly true when fishers employ different fishing methods. Reports of scallop dredgers maliciously or accidentally dragging and damaging static potting gear are not uncommon around the UK (Kaiser et al. 2000). These instances often lead to a series of accusations and counter accusations and can be financially and / or environmentally motivated (Hart 1998). Static fishers often claim that mobile gears destroy their livelihoods by damaging habitat critical to their target species, whereas fishers employing mobile gears accuse the static fleet of denying them access to potential earnings. For example, along the east Yorkshire coast, a gear conflict was recently reported to be underway between static and mobile fishing fleets (BBC 2012). The static fleet, using crab and lobster pots, claimed that scallop boats operating in the area had caused £100,000 worth of damage to their fishing gear over the course of several weeks, and as a result, are currently campaigning to ban scallop dredgers from the area (BBC 2012).

One management mechanism for resolving such gear conflicts is to implement area-based gear restrictions that may operate seasonally or permanently (e.g. the Devon Inshore Potting Agreement - see section 3.1.1). These systems are designed to minimize interactions between incompatible sectors of the fishing industry. In addition, closing areas to scallop dredging can provide considerable benefits to marine biodiversity by protecting non-target species and habitats, and can also increase stock biomass of the species targeted by dredgers (Hart 1998; Kaiser et al. 2000; Blyth et al. 2002; Kaiser 2007b). Considering they both reduce conflict and can be of conservation value, management plans that spatially or temporally separate static and scallop fisheries should be encouraged and become more widespread.

2. The ecosystem effects of scallop dredging in the UK

2.1 Effects on scallop populations

Fishing can have numerous impacts on the species they target and dredging for scallops proves no exception. Some of these impacts are unique to scallop dredging alone, whilst others apply to the many different methods of fishing.

The primary effect of fishing is a reduction in the abundance of target organisms. As scallops reproduce by releasing gametes into the water column (Brand 2006b), reductions in scallop population density can rapidly result in reduced fertilisation success and recruitment (Macleod et al. 1985; Stoner & Ray-Culp 2000; Vause et al. 2007). High levels of fishing can also negatively impact scallop recruitment by truncating age structures. As already explored, minimum legal landing sizes and mesh sizes are employed in the management of scallop fisheries around the UK in order to protect juveniles and allow scallops to spawn at least once before capture. However, because fishing mortality is often high once scallops reach legal size, very few individuals are able to reach the large sizes they would in undisturbed populations (Beukers-Stewart et al. 2005). Not only are larger scallops economically more valuable, they also have more developed reproductive organs capable of producing substantially more eggs (Beukers-Stewart et al. 2005; Kaiser et al. 2007). The removal of scallops before they get to a large size can therefore have a disproportionately high impact on reproductive output and recruitment, threatening the ability of stocks to breed at sustainable levels in the future (Roberts et al. 2005). Age truncation has also been shown to reduce the capacity of populations to buffer environmental events (reviewed in Hsieh et al. 2006).

The physical impacts of towing scallop dredges can further contribute to the unsustainability of scallop fisheries. Firstly, the ability of scallops to swim and escape predators has been shown to be negatively affected by the physical disturbance caused by passing dredges, and by being captured and discarded overboard (Jenkins & Brand 2001). Affected scallops show no signs of recovery even after 24 hours, increasing the predation risk to both those scallops impacted by the dredge and not caught, and those scallops returned to the sea after capture. Secondly, the teeth used on scallop dredges can cause considerable, sometimes fatal, physical damage to the shells of scallops impacted by the passing dredge (Fig. 8; Beukers-Stewart et al. 2001, 2012; Jenkins et al. 2004). In addition to attracting predators and becoming highly susceptible to predation (Jenkins et al. 2004), such physical damage and disturbance can result in reduced levels of growth and reproductive output as metabolic energy is diverted to repairing shell damage which could otherwise be invested in growth and gonad development (Beukers-Stewart et al. 2005; Kaiser et al. 2007).

Due to their penetrative nature and close contact with the seabed, scallop dredges cause substantial physical disruption to the seafloor by ploughing sediments and damaging organisms attached to or resting upon seabed, such as hydroids, bryozoans, sponges and maerl (Dayton et al. 1995; Jennings & Kaiser 1998; Kaiser et al. 2000). In addition to dramatically reducing an area's capacity to support other biodiversity (see section 2.2.3), the removal of such organisms is known to have severe consequences on scallop recruitment as they provide essential habitat for the settlement of scallops and other invertebrates (Fig. 9; Bradshaw et al. 2001; Kamenos et al. 2004a). Consequently, such locations are often referred to as "nursery areas" as they tend to be highly productive, support high levels of juvenile density, growth and survival, and contribute disproportionally to the production of adult recruits (Beck et al. 2001; Gibb et al. 2007; Laurel et al. 2009). The damage inflicted upon nursery habitats by fishing gears has therefore been shown to negatively impact scallop recruitment (Collie et al. 1997; Bradshaw et al. 2002), whilst the protection of nursery habitats has been shown to dramatically enhance scallop settlement levels (see section 3.2.3).

Figure 8 | Fatal damage to king scallops that were impacted by a scallop dredge but not captured. The damage caused by the teeth of the scallop dredge is particularly noticeable. *Photo: Howard Wood*

By increasing mortality and reducing recruitment, the impacts mentioned above all have the potential to negatively affect the long-term sustainability of scallop fisheries in the UK. Scallop stocks located around Scotland currently account for over half of the UK king scallop fishery (Dobby et al. 2012) but concerns have recently been raised over increasing mortality, declining recruitment and spawning stock biomass in several major Scottish stocks (Hall-Spencer & Moore 2000; Howell et al. 2006; Hinz et al. 2011; Barreto & Bailey 2013). These problems are not unique. Scallop fisheries are well known for exhibiting dramatic fluctuations in recruitment, landings and abundance (Paulet et al. 1988; Orensanz et al. 1991; Beukers-Stewart et al. 2003; Beukers-Stewart & Beukers-Stewart 2009). Such fluctuations are difficult to incorporate into fisheries management and can result in their sudden and unexpected collapse (Frank & Brickman 2001; Beukers-Stewart & Beukers-Stewart 2009). Furthermore, recruitment and mortality of scallop stocks are predicted to become increasingly more erratic in the future due to ocean acidification (Gazeau *et al*. 2007, Kurihara 2008, Watson *et al*. 2009). This is being caused by increased ocean uptake of anthropogenic carbon dioxide; a process which reduces the amount of carbonate available to scallops to form their protective shells (Sabine et al. 2004; Doney et al. 2009). Ocean acidification is expected to affect the early history stages of scallops most dramatically by reducing shell growth and increasing mortality (Andersen et al. 2013). Weaker shells also make juvenile and adult scallops more vulnerable to predation and damage from fishing gears (Beukers-Stewart et al. 2012). Ocean acidity is currently increasing at a rate unprecedented for tens of millions of years (Doney et al. 2009), meaning scallop fisheries all over the world are badly exposed to risk if the species they target cannot adapt. Stronger efforts must therefore be made to safeguard the long-term sustainability of commercially important scallop stocks in the UK whilst reducing the environmental impact of their fisheries.

Figure 9 | Juvenile scallops preferentially settle in structurally complex habitats, such as kelp stipes and macroalgal fronds (top images - *Photo: Angus Robson*), and bryozoans and hydroids (bottom image - *Photo: Hilmar Hinz*). Their damage and removal has therefore been shown to negatively impact scallop recruitment.

2.2. Effects on marine ecosystems

Of all the fishing gears, scallop dredges are considered to be the most damaging to non-target benthic communities and seafloor habitats (Collie et al. 2000; Kaiser et al. 2006). Furthermore, the Newhaven dredges used by the UK king scallop fishery are likely to be one of the most damaging types of scallop dredge due to the effect of their long teeth, which can penetrate 3-10 cm into the seabed (Beukers-Stewart & Beukers-Stewart 2009; Shephard et al. 2009; Craven et al. 2013). Given the recent and rapid expansion of the UK scallop fishery, this is of particular concern to fisheries managers and conservation scientists (Shephard et al. 2010; Craven et al. 2013).

The effects of scallop dredging on marine ecosystems vary with different seabed types, levels of background disturbance, local hydrography, fishing intensity and the characteristics of the ecological community (Kaiser et al. 1996; Auster et al. 1996; Bradshaw et al. 2001). However, the following sections address scallop dredging impacts that generally apply to any marine ecosystem.

2.2.1. Physical impacts

Dredges are specifically designed to penetrate and disrupt surface sediments in order to increase the catch rate of the scallops. In doing so, scallop dredging can bring about a number of physical alterations to the seabed and surrounding environment.

Overall, the general effect is that they cause homogenization of sediments and topography through penetration, mixing and flattening of sediments (Collie et al. 2000). Natural seabed features such as ripples, pits and burrows can all be eliminated by scallop dredging. In their place, dredging sculpts the sediment into 3 cm high ridges which can persist for up to three years in low wave / tide energy environments (Hall-Spencer & Moore 2000). Scallop dredging can also move and / or remove significant quantities of stones and boulders from fishing grounds (Eleftheriou & Robertson 1992; Bradshaw et al. 2002) which has been reported to cause shifts in the granulometric structure of surface sediments (Hall-Spencer & Moore 2000). Any changes in sediment topography will likely alter near bed hydrodynamics, which can result in the deposition of fine sediments (Probert 1984; Dernie et al. 2003). In addition, the removal or disturbance of surface sediments can change patterns of nutrient cycling or carbon flux, for example, by exposing underlying anaerobic sediments (Watling et al. 2001; Kaiser et al. 2002).

The disturbance caused by dredges can also re-suspend soft sediments, nutrients, eggs, cysts and small organisms buried into the sediment (O'Neill et al. 2013). Particular concerns have been raised about this as high levels of suspended sediment can smother surrounding sessile marine life, burying important habitats such as maerl (see section 2.3.1) and clogging the feeding and respiratory organs of filter feeding organisms, such as mussels and scallops, thereby impacting on their reproduction (Brand 2006b; Dale et al. 2011; Szostek et al. 2013).

2.2.2. Burrowing infauna

As scallop dredges can penetrate anywhere between 3-10 cm into the seabed (Kaiser et al. 1996) they have a strong potential to disrupt the benthic infauna; the organisms that burrow and live within the sediment. Any impact scallop dredging has upon the infauna can percolate through the entire marine ecosystem as they constitute an important food resource to fish, invertebrates and other higher trophic levels (Daan et al. 1990). The benthic infauna also play a significant role in linking benthic and pelagic processes by transferring energy to pelagic organisms derived from primary production and falling detritus (Newell et al. 1998), in addition to influencing the structure of planktonic food webs (ICES 2001). Furthermore, burrowing species of infauna often play a key role in controlling the scale and direction of nitrogen flux in benthic communities (Leslie & Shelmerdine 2007). However, by destroying the burrows of infaunal organisms and favoring the growth of small, highly abundant burrowing species over less common larger ones, fishing disturbance can alter the rate of nutrient flux (Kaiser et al. 2002, Leslie & Shelmerdine 2007).

Compared to other taxa, little is known about the effects of scallop dredging on benthic infauna. Generally, it is thought a proportion of the benthic infauna will detect and react to an oncoming scallop dredge by entering the water column or burrowing deeper into the sediment. However, those that remain in the sediment will be subject to the same physical forces as the sediment they inhabit, meaning they may become crushed or suspended in the water along with the sediment (O'Neill et al. 2013). The fate of infaunal organisms will depend on the damage and stress they sustain, where they resettle and whether they are at an increased risk of predation. O'Neill *et al* (2013) found no difference in the abundance and biodiversity of infauna between dredged and undredged sites on the west coast of Scotland, whereas a study off the Isle of Man found that dredging significantly reduced the biomass of infauna (Kaiser et al. 2000). Studies conducted outside the UK also report inconsistent results of dredging on infaunal communities, with some observing no effect, and others reporting strong changes to infaunal abundance and biodiversity that can persist for 8-12 months (reviewed in Løkkeborg 2005).

2.2.3. Epifaunal and sessile organisms

In theory, some mobile organisms should be able to detect an oncoming scallop dredge and move out of its way, enter the water column or burrow deep into the sediment, thereby avoiding damage and / or capture. However, this response is not possible for the benthic epifauna, the organisms attached to the seabed, making them particularly vulnerable to scallop dredging (Ramsay & Kaiser 1998).

Organisms that attach to the seabed are functionally important to marine ecosystems as they provide an element of 3-dimensional structure to often otherwise featureless seafloors. In doing so, they supply important refuges for small / juvenile fish from predators and unfavourable environmental conditions (Monteiro et al. 2002; Ryer et al. 2004; Cacabelos et al. 2010), represent important feeding sites for fish and invertebrates (Bradshaw et al. 2003; Warren et al. 2010), and provide essential habitat for the settlement of scallop spat and a range of other organisms, including the settlement of further epifauna (Howarth et al. 2011). Upright hydroids, for example, have been found to provide an attachment surface for scallops, nudibranchs, bryozoans, barnacles, sponges, tube-dwelling worms and other hydroids (Bradshaw et al. 2001). Such locations are therefore often referred to as nursery areas as they tend to be highly productive, support high levels of juvenile density, growth and survival, and contribute disproportionally to the production of adult recruits (Beck et al. 2001; Gibb et al. 2007; Laurel et al. 2009b). Commonly cited nursery areas include maerl beds (see section 2.3.1; Kamenos et al. 2004b, 2004a; Hall-Spencer et al. 2006), seagrass beds (Warren et al. 2010) and areas of dense macrophytes / macroalgae (Christie et al. 2007; Cacabelos et al. 2010; Howarth et al. 2011), all of which have been shown to harbour high densities of commercially exploited species such as spider crabs, *Maja squniado,* juvenile cod, *Gadus morhua*, edible crab, *Cancer pagurus* and edible sea urchins, *Echinus esculentus*. In addition, many epifaunal species support unique micro-communities e.g. caprellid amphipods on hydroids, the range of invertebrates associated with kelp forests, or the diversity of organisms associated with pomatocerid tube worm heads (Kaiser et al. 1999; Airoldi et al. 2008). Consequently, the removal / damage inflicted on nursery habitats from towed fishing gears can create a series of knock-on effects, reducing an area's capacity to biodiversity and negatively impacting upon the recruitment of commercially important species (Collie et al. 1997; Bradshaw et al. 2001, 2003; Kaiser et al. 2005).

Long-lived, slow-growing, upright epifaunal species often have fragile body structures and are especially sensitive to encounters with fishing gear, whereas smaller taxa are more resilient (Kaiser et al. 2000; Hall-Spencer & Moore 2000). For example, slow growing sponges and soft corals take much longer to recover (up to 8 years) from scallop dredging than organisms with shorter life-spans such as polychaete worms and encrusting bryozoans (less than 1 year; Kaiser et al. 2006). Hence, experimental dredging conducted in the Irish Sea was found to shift the benthic community from one state to another, going from a community dominated by upright species to one dominated by small, encrusting, opportunistic, fast growing species that offered much less 3-dimensional structure (Bradshaw et al. 2001). Similarly, a later study on several fishing grounds in the Irish Sea found scallop dredging to reduce the overall biomass of the epifaunal community and for the community to become dominated by smaller-bodied organisms (Lambert et al. 2011). By destroying epifaunal assemblages, scallop dredging can cause a reduction in the range of ecological niches available for associated biodiversity that rely on epifaunal organisms for complex habitat, shelter and food (Auster et al. 1996; Collie et al. 1997; Bradshaw et al. 2003; Lambert et al. 2011).

2.2.4. Mobile species

In addition to capturing scallops, the dredges used by the UK scallop fishery capture a wide variety of non-target mobile megafauna, including some commercially important species (Fig. 10). Examples include: fish (flatfish, dog fish, skates, rays, monkfish and dragonets), crustaceans (edible crabs, swimmer crabs, spider crabs and hermit crabs), echinoderms (brittlestars, starfish and sea urchins), molluscs (bivalves and gastropods), and cephalopods (octopus and cuttlefish; Bradshaw et al. 2001; Craven et al. 2012). Although scallop dredges are considered to be relatively "clean" compared to other types of mobile fishing gear such as beam trawls (Kaiser 2007b), a study off the Isle of Man found that for every scallop captured by a Newhaven dredge, four individuals of by-catch were also caught (Hinz et al. 2012). Commercially valuable species are retained in some cases, particularly edible crabs and monkfish in the Isle of Man dredge fishery (Beukers-Stewart et al. 2001; Brown 2013; Craven et al. 2013) and cuttlefish in the English Channel dredge fishery (Enever et al. 2007) but the majority of by-catch is discarded damaged, dying or dead (Beukers-Stewart et al. 2001; Jenkins et al. 2001).

An assessment of the 10 most common by-catch species in the Irish Sea scallop fishery found that approximately 20 to 30 % of individuals suffered fatal damage after dredge capture (Shephard et al. 2009). However, the proportion of individuals impacted by dredging varies greatly between species, even within the same family (e.g. starfish). The most sensitive species to dredge damage include the seven armed starfish, *Luidia ciliaris*, the edible sea urchin, and the commercially important edible crab (Jenkins et al. 2001; Veale et al. 2001). In contrast, the pin cushion starfish, *Porania pulvillus*, rarely appears to suffer any damage from being captured in a scallop dredge (Beukers-Stewart et al. 2001; Jenkins et al. 2001). Initial contact with the dredge teeth appears to cause most of the fatal damage suffered by by-catch species, while non-fatal damage appears to occur in the mesh bag during the tow and landing of the catch (Shephard et al. 2009). Intensity of non-fatal damage may be related to the amount of stones in the catch and the fullness of dredges – suggesting shorter tow lengths could reduce this type of damage (Bradshaw et al. 2001).

Figure 10 | A large monkfish (*Lophius piscatorious*) captured in a scallop dredge while fishing around the Isle of Man. Monkfish routinely suffer serious damage when captured in scallop dredges (*Photo: Bryce Stewart*)

Compared to crustaceans and starfish, a relatively small number of fish are caught by scallop dredges. A study in the Irish Sea recorded that 97.6% of tows of scallop gear generated fish by-catch belonging to 50 different species, of which the majority were monkfish (Fig. 10; Craven et al. 2013). However, relative to the target species, fish by-catch was low; estimated at 1 fish per 103 scallops captured. Then again, when entire scallop fleets are considered, the number of fish removed can be quite substantial; estimated to be 3.3 million fish per year by the English Channel scallop fleet (Enerver et al. 2007). There is also evidence that by-catch in the Isle of Man scallop dredge fleet was at least partially responsible for a decline in monkfish over a 14 year period (Craven et al. 2013).

High levels of mortality may also occur in organisms that are impacted by the scallop dredge but not necessarily captured. Through the use of SCUBA surveys, Jenkins *et al*. (2001) found that over 75% of the megafauna which encountered scallop dredges remained on the seafloor. These organisms displayed surprisingly similar levels of damage and mortality as the by-catch landed on deck, which was caused by crushing as animals passed around, through or under the heavy gear, or by the initial encounter with the tooth bar. They also found that damage to commercially valuable edible crabs was highest when they had been impacted by the dredge but not caught. An associated study found that dredging resulted in the capture of approximately 25% of the edible crabs present in the dredge path, but that more than 40% of the remaining crabs were left dead or dying on the seabed (Beukers-Stewart et al. 2001). Scallop dredging is therefore a very inefficient way to catch crabs, and wastes a resource that would be otherwise available to fishermen employing static gears. Furthermore, towed fishing gears can cause entanglement and loss of crab pots when operating in the same area as crab fisheries. These patterns can give rise to considerable conflict between crab and scallop fisheries (see section 1.7). Management plans that spatially or temporally separate scallop and crab fisheries (e.g. the Devon Inshore Potting Agreement) should therefore be encouraged (see section 3.1.1).

Paradoxically, some organisms are attracted to areas that have been scallop dredged and consequently increase in abundance. A study in the Irish Sea found the densities of scavengers and predators, such as starfish, crabs and dog fish, to increase by up to 200 times in the presence of scallop fishery discards (Veale et al. 2000). Scavengers are also attracted to the disturbed sediment, and to the damaged or dead organisms left behind by the wake of the dredge (ICES 1992). High densities of scavengers can result in elevated predation pressure on some organisms (Ramsay & Kaiser 1998) particularly where some individuals have already been damaged by fishing activity (Veale et al. 2000; Jenkins et al. 2004). This may place added predation pressure on other organisms in the area, including scallops and other commercially targeted species (Beukers-Stewart & Beukers-Stewart 2009). In addition, being lured to fishing grounds may place these species at increased risk of being caught or damaged during the next pass of the fishing gear (Bradshaw et al. 2000). However, due to dispersion of odour plumes, sediment resettlement and predation of damaged organisms, the high densities of scavengers gathering at dredged grounds is likely to be a short-lived event. Then again, a broad-scale study by Bradshaw et al. (2002) in the Irish Sea found that mobile, robust, and scavenging invertebrate species had increased in abundance over a 60 year time period while slow-moving or sessile, fragile taxa had decreased. Likewise, a study in the Isle of Man found that the density of scavenging dog fish significantly increased over a 14 period whereas the density of commercially important monkfish decreased (Craven et al. 2013). Both dog fish and monkfish were caught in substantial numbers but differed in their post-discard survival. Dog fish have remarkably high post-discard survival rates of up to 98% (Rodríquez-Cabello et al. 2005) which may be why they were not negatively affected by high levels of fishing disturbance, whereas monkfish routinely suffer serious damage when captured in scallop dredges, and grow and reproduce slowly, meaning they are much more vulnerable to depletion by scallop dredging (Craven et al. 2013).

Overall, most studies indicate that benthic communities in areas subject to a long history of scallop fishing will have become simplified to a suite of species that are relatively resistant to fishing disturbance (Currie & Parry 1996; Bradshaw et al. 2002; Brown 2013). This can make it difficult to detect the effects of fishing within contemporary benthic communities. Within these altered ecosystems, normal levels of fishing may have relatively little effect on community structure. For example, a recent analysis of 15 years of data on mobile benthic invertebrate species around the Isle of Man found that fishing pressure only had a small negative effect on patterns of diversity (Brown 2013). More detailed examination of individual species indicated that the abundance of the dredge resistant common starfish (*A. rubens*) and cushion stars (*P. pulvillus*) was more strongly influenced by environmental factors (chlorophyll-α and temperature) than fishing disturbance (Brown 2013). However, when an area of seabed around the Isle of Man was protected from fishing, the overall density of benthic species, especially king scallops and edible crabs, recovered dramatically (Bradshaw et al 2001; Beukers-Stewart et al 2005, Brown 2013). In contrast, the density of the scavenging common starfish declined significantly over time within the protected area (Brown 2013, see section 3.2.1).

2.3. Effects on different seabed types

As discussed, the effects of scallop dredging vary with different seabed types, level of background disturbance, local hydrography, fishing intensity and the characteristics of the ecological community (Kaiser et al. 1996; Auster et al. 1996; Bradshaw et al. 2001). The following sections address these differences.

2.3.1. Maerl

Maerl (Rhodophyta: Corallinaceae) is a red algae that forms hard, brittle, filaments made of calcium carbonate (Fig. 11). It can accumulate to form deep, loose lying beds that can cover anywhere between 10 m^2 to several $1000m^2$ (Kamenos et al. 2004b, 2004a). Maerl beds are structurally very complex, and as a result, often support tremendous levels of biodiversity (Birkett et al. 1998; Hall-Spencer & Moore 2000; Kamenos et al. 2004b) as well as high densities of juvenile scallops, cod and edible crab, all species of commercial interest in the UK (Hall-Spencer et al. 2008). They are therefore listed as a UK Biodiversity Action Plan (UKBAP) priority habitat, in Annex I of the EU Habitats Directive, as a threatened and/or declining species under the Oslo and Paris (OSPAR) Habitats Convention for the Protection of the Marine Environment of the North-East Atlantic, as well as being subject to a number international conservation legislation provisions (www.naturalengland.org.uk).

Figure 11 | Maerl beds can provide the seabed with very high levels of structural complexity and can support extraordinary levels of biodiversity. Image taken from: www.algosophette.com

Maerl beds are usually characterized by coarse sediment, clear water, and strong currents (to prevent smothering by silt), and thus often provide good scallop fishing grounds (ICES 1992). However, maerl beds are fragile and very slow growing, often taking thousands of years to build up, meaning they are exceptionally vulnerable to damage by scallop dredging (Giraud & Cabioch 1976; Foster 2001; Grall & Hall-Spencer 2003). A single impaction event with a scallop dredge can significantly reduce the structural complexity of a maerl bed by breakage, and can kill the maerl by burying it under sediment (Hall-Spencer & Moore 2000; Kamenos et al. 2003). For example, a study off the west coast of Scotland found that a single tow of three scallop dredges crushed and compacted maerl beds and buried the maerl 8 cm below the sediment surface (Hall-Spencer & Moore 2000). The passing of the dredge also caused resuspension of sediments which blanketed an area at least 12 times the area that had experienced contact with the gear, reducing the maerl's ability to photosynthesize. These combined effects led to a 70-80% reduction in live maerl, which displayed no signs of recovery even after four years. It was concluded that the lack of recovery of was related to the slow growth and poor recruitment of maerl. In reality, the effects of scallop dredging on maerl beds are likely to be even stronger as scallop dredgers often tow many more dredges than the three utilised in the above study, and fishers are likely to repeatedly dredge an area several times due to gear inefficiency (Beukers-Stewart et al. 2001). Losses to maerl beds in the UK will substantially reduce regional biodiversity and can impact commercial fisheries by diminishing nursery-area function (Kamenos et al. 2004b).

2.3.2. Modiolous reefs

Similar to maerl, horse mussels (*Modiolus modiolus*) can accumulate in large, dense aggregations, forming distinctive biogenic habitats rising up to 3 m above the surrounding seabed, known as *Modiolus* reefs (Fig. 12; Wildish et al. 1998). *Modiolus* reefs are regarded as ecosystem engineers as the mussels that form the reef bind to each other, and to the seabed, with byssal threads, which has a stabilising effect on the seabed (Rees 2009). In addition, through binding living mussels, dead shell and fine sediments, they alter both the topography and the sediment composition of the seabed in and around the reef (DOE 2005; Ragnarsson & Burgos 2012). The biological activity of the mussels themselves also affect ecosystem functioning by filtering large volumes of seawater and altering nutrient fluxes (Hargrave et al. 2008; Callier et al. 2009; Dolmer & Stenalt 2010). *Modiolus* reefs can support high levels of biodiversity as they provide a hard surface for the attachment of algae, kelp, sponges, hydroids and soft corals (Rees 2009). In addition, the mussel matrix itself can support a rich community of crevice-dwelling infauna of 200-300 species at densities exceeding 22,000 individuals per m^2 (Ragnarsson & Raffaelli 1999; Sanderson et al. 2008). *Modiolus* reefs have therefore been identified as rare biodiversity hotspots and are listed in Annex I of the EU Habitats Directive, as a threatened and/or declining species by OSPAR, and a UKBAP priority habitat.

Although a widespread and common species around the UK, true *Modiolus* reefs are restricted to small areas around the Isle of Man, Irish Sea and Scotland (DOE 2005). Due their long life span (over 48 years), slow growth and poor recruitment, *Modiolus* reefs have been identified as particularly vulnerable to the physical impacts of fishing (Cook et al. 2013). For example, a substantial *Modiolus* reef was previously located south off the Isle of Man but was eliminated by intensive scallop dredging in the 1970s and 1980s (Rees 2009). Similarly, in Strangford Lough, Northern Ireland, *Modiolus* reefs that used to cover extensive areas were reduced to isolated small clumps by scallop fishing (Rees 2009). In addition to flattening and killing *Modiolus* reefs and destabilising the seabed, a recent study found that experimentally scallop dredging *Modiolus* reefs off the Isle of Man and Wales reduced the biodiversity of the associated community by 59-90% (Cook et al. 2013). No signs of recovery were detected a year later, and given the life history recovery of horse mussels, recovery could take many years.

Figure 12 | Horse mussel reefs often support exceptionally high levels of structural complexity and biodiversity (*Photo: Richard Shucksmith*).

Given the importance of biogenic reefs (such as maerl and *Modiolus* reefs) to both fisheries and biodiversity, along with their inherent vulnerability to disturbance, there is a strong argument for completely protecting biogenic reefs from all towed fishing gear, at the very least.

2.3.3. Soft sediments

Although scallop fisheries are known to have negative impacts in almost all habitat types, some are highly sensitive to disturbance while others are more resilient. In general, the more naturally stable an area of seabed is, the more sensitive the ecological community will be to disturbance (Eleftheriou & Robertson 1992; Collie et al. 2000). It is therefore commonly thought that the effects of dredging will be relatively short-lived for ecological communities adapted to frequent natural disturbance by currents, tides, storms and re-suspension of sediment, such as those inhabiting soft mud / sand sediments (ICES 1992; Jennings & Kaiser 1998; Collie et al. 2000; Dernie et al. 2003; Sciberras et al. 2013). However, mixed sand and mud habitats often support diverse benthic communities of high biomass and tend to be the main focus of commercial scallop fisheries in the UK (Bradshaw et al.

2000; Kaiser et al. 2006). It is therefore important to understand how dredging can affect communities living in these sediments.

A 2 km^2 area off the Isle of Man was protected from fishing in 1989 (See section 3.2.1). Bradshaw *et al*. (2001) began experimentally dredging a section of the closed area five years later to see how communities in mud and sand habitats reacted. It was found that scallop dredging caused the benthic community to change from one state to another, going from a community dominated by upright species to one dominated by small, fast growing encrusting species that offered much less 3-dimensional structure. Such changes have been observed by other studies and can cause a series of knock on effects that dramatically reduce marine biodiversity (Collie et al. 1997; Watling & Norse 1998; Kaiser et al. 2000; Bradshaw et al. 2003). Similarly, with the aim of assessing the effects of scallop dredging on the benthic community, experimental dredging was carried out in a small sandy bay off the east coast Scotland (Eleftheriou & Robertson 1992). The area was identified as high-energy, being both shallow and strongly exposed to wave action. Consequently, the infaunal community showed little response to scallop dredging. However, large numbers of molluscs, star fish, sea urchins, crabs and sand eels were killed and damaged by scallop dredging. Finally, Sciberras et al. (2013) examined temporal changes in scallop density and epibenthic communities over 23 months at two areas (one closed to fishing and one open seasonally) also in a dynamic area of seabed, this time in the Cardigan Bay SAC, Wales. No significant differences were found between the two areas, with changes in abundance of both scallops and epifauna appearing to be driven seasonal fluctuations rather than any form of recovery within the closed area. This led the authors to conclude that natural disturbance was a more important structuring factor than fishing at this site. However, if the experiment did not detect an effect of the fishery on the target species (king scallops), which by definition must have removed some individuals, then fishing must have been at a fairly low level during the study period. Alternately / in addition, illegal fishing in the protected area, which has subsequently been revealed as a major problem (www.milfordmercury.co.uk/news/9666273.Court_dishes_out___29_000_in_fines_for_illegal_scalloping/;www.westerntelegraph.co.uk/news/county/11084521.Illegal_Cardigan_Bay_scallop_dredgers_face_fine_of_up_to___1m_in_biggest_case_ever_brought_by_Welsh_Government/?ref=var_0) may have reduced any differences between the open and closed sites, although this was not apparent in VMS records of fishing activity during the study (Sciberras et al. 2013)

Unlike biogenic habitats such as maerl and *Modiolus* reefs, benthic communities in softer sediments will recover if protected from fishing, although time frames vary for different species (Bradshaw et al. 2003; Kaiser et al. 2006). Surveys of the Port Erin closed area off the Isle of Man (see section 3.2.1; Brown 2013) demonstrated that the densities of some species (e.g. king scallops and edible crabs) were still increasing even after 17 years of protection.

2.3.4. Rocky reefs and mixed substrates

The spring action of Newhaven scallop dredges mean they are unlikely to perform well on hard and uneven grounds, and damage to fishing gear may render such grounds unprofitable. Consequently, maps of fishing effort indicate scallop dredgers tend to avoid areas of rocky reefs, boulders and bedrock slabs. Nonetheless, it is a possibility that rocky-reef habitats suffer some damage from scallop dredging activity since they are often found in close proximity to commercially viable scallop grounds (Boulcott & Howell 2011). Dredging performed on mudstone reefs in Lyme Bay, southern England, reduced the abundance and size of structurally complex bryozoans, soft coral and sponges by 54-73% compared to unfished sites (Hinz et al. 2011). Similarly, a study on rocky reefs in the sound of Jura, west Scotland, also found dredging to damage bryozoans, hydroids, soft corals and sponges, but that the damage was incremental, increasing with the number of dredge tows performed (Boulcott & Howell 2011). This is in contrast to softer sediment habitats where the majority of damage occurs on the first tow of fishing gear through a pristine site (Kaiser et al. 1996; Collie et al. 2000). The authors therefore concluded that there is considerable value in protecting slow growing, rocky-reef communities even if they have experienced low to moderate fishing in the past. Likewise a further study in Scotland on mixed substrates (Boulcott et al. 2014) found that the emergent fauna on hard substrates which are suited to dredging, such as pebbles and cobbles, are particularly vulnerable to dredging and should be protected.

3. Managing the effects of scallop dredging in the UK

3.1. Case studies of successful management of scallop fisheries in the UK that resolve conflict

Considering the conflicts and large environmental impacts associated with scallop dredging, there is an urgent need for better management of scallop fisheries in the UK. In many areas around the UK, conflict has arisen between different sectors of the fishing industry that operate within the same area. Reports of scallop dredgers maliciously or accidentally dragging and damaging static potting gear are not uncommon (Kaiser et al. 2000; BBC 2012). The following section shows how setting aside different areas for different fisheries has proved a successful management strategy for reducing gear conflicts in the UK, and one which can also generate unexpected conservation benefits.

3.1.1. South Devon Inshore Potting Agreement (IPA), England

Several different fisheries operating different gears work the south coast of Devon, England. Static fishermen using static pots target edible crabs and European lobsters (*Homarus gammarus*), scallop dredgers target king scallops, and beam and otter trawlers target plaice (*Pleuronectes platessa*) and sole (*Solea solea*). The static fishermen operating in the area often deploy their gear and leave it unattended on the seafloor for anywhere between 24-72 hours (Hart 1998; Kaiser et al. 2000). The number of crab pots in the area was often so high that they frequently became tangled in towed fishing gear, causing loss of gear and earnings in the static sector (Kaiser et al. 2000). Eventually the interference became a serious problem and a management system was required to partition the fishing grounds, thereby minimising contact between the two types of gear (Hart 1998).

A series of voluntary gear restrictions were introduced in South Devon in 1978, and later became statutory in 2002 (Blyth et al. 2002). This Inshore Potting Agreement (IPA) covers a number of areas totalling 478 km^2, of which 350 km^2 are reserved for the use of static gears only (Blyth et al. 2002). The voluntary agreement exists between several different parties represented by the South Devon and Channel Fishermen Ltd. (pot fishers), the Trawler Owners Association (towed gears), and the Devon and Severn Inshore Fisheries and Conservation Authority (local enforcement agency – previously the Devon Sea Fisheries Committee). Some areas are open to towed gears at certain times of the year, whilst others are not.

Initially concerns were raised about compliance. As stock biomass would be expected to become more abundant in the areas closed to towed gears, it was thought that fishers may be tempted to enter the closed areas in order to obtain higher catch rates. If some towed fishermen did break the agreement, it would undoubtedly have a destabilising effect on the IPA (Hart 1998). Despite this, the partitioning system off Devon has remained stable for over 35 years.

The south Devon IPA is widely regarded as a success by both fishers and managers because it has effectively allowed fishers from both sectors to operate profitably on traditional fishing grounds (Blyth et al. 2002). An unplanned for, but welcome side effect of this agreement has been considerable benefits to marine biodiversity in the areas where towed gears have been excluded

(Kaiser et al. 2000; Blyth et al. 2002, 2004). Responses have included significant increases in the biomass of hydroids, soft corals and other important nursery habitats, as well as increases in long-lived molluscs (*Glycymeris glycymeris*) and large burrowing urchins (*Spatangus purpureus*). These two species are particularly vulnerable to fishing because they live close to the sediment surface, reproduce infrequently (*G. glycymeris*) and have fragile shells that are damaged easily by physical contact with towed fishing gears (Kaiser et al. 2000). Scallop densities have also increased within the areas closed to towed gears, potentially increasing scallop recruitment both inside and outside the protected areas, as well as a number of fish species which have also increased in abundance (Blyth-Skyrme et al. 2006).

Several factors have been identified as critical to the success of the south Devon IPA including (Blyth et al. 2004; Blyth-Skyrme et al. 2006):

- It was formed collectively by fishers and not a regulating authority.
- A small number of organisations represented fishers.
- Those organisations had very high levels of membership.
- The management system was simple to implement, explain and understand.

However, there are some problems that need addressing, including conflict within the static sector. Within the south Devon IPA, occupation of an area traditionally signifies the right to fish in that location, but only as long as the gear remains there. The density of fish pots in the IPA is so high that space for new static gear is limited, and fishers wishing to enter the static gear fishery are unable to do so unless they buy second-hand gear already positioned at sea. Vacant sites are also limited because some fishers leave weighted marker buoys in place, falsely signifying that an area has been taken, thereby discouraging other fishers from setting theirs pots in areas which are, in reality, unoccupied. As territories cannot be expanded, fishers can only create space for additional pots by moving their existing gear closer together (Blyth et al. 2002). At the nearby Lyme Bay MPA, where similar issues were arising, and innovative new project is addressing the problem (see below).

3.2. Case studies of successful management of scallop fisheries in the UK that address their environmental impacts

Following a large number of recently established policies and initiatives, closing areas to some or all types of fishing through the implementation of marine protected areas (MPAs) and marine reserves is likely to increase in the UK over the next few decades. The EU Marine Strategy Frameworks Directive (MSFD), Birds and Habitats Directives, OSPAR, HELCOM (Helsinki Commission) and Barcelona regional seas conventions, have all initiated the process of establishing a coherent network of MPAs within European waters (Fenberg et al. 2012; Metcalfe et al. 2013). On a national level, the planned implementation of Marine Conservation Zones (MCZs; England, Wales and Northern Ireland) and Scottish MPAs (Scotland) (see above) will all lead to the creation of a network of MPAs around the United Kingdom (Jones 2012, JNCC 2013). All these measures intend to achieve a variety of management goals; principally to conserve biodiversity and promote the sustainability of fisheries (Pomeroy et al. 2005; Metcalfe et al. 2013).

The establishment of Marine Protected Areas (MPAs) and marine reserves is supported by a growing number of scientific studies that have shown that closed area protection can increase the abundance and mean size of target species (Halpern & Warner 2002; Halpern 2003; Lester et al. 2009), enhance local reproductive output (Roberts et al. 2001; Gaines et al. 2003; Grantham et al. 2003) and improve the survival and growth of juveniles (Myers et al. 2000; Beukers-Stewart et al. 2005). All of these effects may then result in the greater production of larvae, juveniles and adults which then disperse ("spillover") to grounds outside the closed area and contribute to fishery landings (McClanahan & Mangi 2000; Pelc et al. 2010). Closing areas to destructive fishing methods can also allow seafloor habitats to recover, enhancing biodiversity and generating a number of benefits that flow back to commercially targeted species (Jennings & Kaiser 1998; Howarth et al. 2011). It is these ideas that underlie the current push towards 'ecosystem-based fishery management', where management priorities begin with the ecosystem, moving away from traditional single-species focussed approaches (Pikitch et al. 2004; Zhou et al. 2010).

The implementation of MPAs in Europe is still at a very early stage (Fenberg et al. 2012; Metcalfe et al. 2013) and their use as an ecosystem-based fishery management tool remains a highly contentious issue (Boersma and Parrish, 1999; Jones, 2007; Kaiser, 2004, 2005; Sciberras et al., 2013). However, the following case studies highlight the number of benefits closing areas to towed gears can generate. In many cases, these changes have not only benefited conservation, but also both static and mobile fishing fleets, potentially outweighing the cost of losing access to some fishing grounds.

3.2.1. The Port Erin Closed Area, Isle of Man

Scallop fisheries have existed around the Isle of Man since 1937, and together, the fisheries for king and queen scallops are now by far the most valuable on the island. The king scallop fishery has been subject to a closed season and minimum sizes since the 1940s. However, by the late 1980s, stocks appeared to be in decline and a series of additional management measures started to be introduced. In 1989, a small 2 km^2 area was closed to fishing with mobile gear (and taking of scallops by any means) off Port Erin in the south west of the island to monitor the response of the benthic community in the absence of fishing. Although recovery of the king scallop population was slow at first (at least partly due to illegal fishing in the closed area), it accelerated over time (Fig. 13). After seventeen years of protection, king scallop densities were thirty times greater within the closed area than when first protected (Beukers-Stewart et al. 2005; Beukers-Stewart & Brand 2007). The reduction in fishing mortality also allowed individuals within the closed area to reach much older and larger sizes, with exploitable and reproductive biomass of scallops being 20 and 33 times higher respectively, than on the adjacent fishing ground by 2006. There is growing evidence that export of larval scallops from high rates of breeding in this closed area has boosted surrounding populations and therefore the fishery (Beukers-Stewart et al. 2004; Beukers-Stewart et al. 2005; Beukers-Stewart & Brand 2007; Neill & Kaiser 2008). Overall, scallop catch rates had recovered to reach a 20 year high on many fishing grounds by the mid-2000s, despite the local fleet being half the size it was in the early 1980s (Beukers-Stewart et al. 2003, Brand et al. 2005).

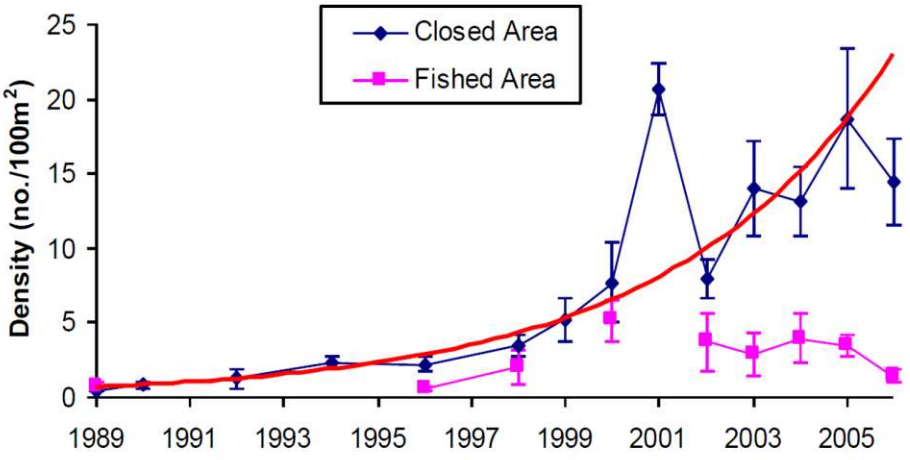

Figure 13 | Mean density (number per 100 m^2) of king scallops in the Port Erin closed area and on the adjacent Bradda Inshore fishing ground off the Isle of Man. Taken from Beukers-Stewart & Brand (2007).

Not only does the Port Erin closed area appear to have helped king scallop populations recover, it has also led to the development of more heterogeneous and structurally complex epibenthic communities, particularly in terms of upright hydroids and bryozoans (Bradshaw et al 2001; 2003). Furthermore, there has been a general increase in the total density of mobile benthic invertebrates over time (Fig. 14; Brown 2013). In other areas, such as on the Georges Bank off the east coast of the USA, increases in scallop predators within protected areas is thought to be a threat to scallop populations (Marino et al. 2007). However, in the Port Erin closed area, densities of the main scallop predator, the common starfish *Asterias rubens*, surprisingly decreased throughout the study period (Brown 2013). This may have been due to changing environmental conditions, or its attraction (as a scavenger) to feed instead on dredge-damaged marine life on nearby fishing grounds (Brown 2013). Either way, low levels of predation pressure within the Port Erin closed area have probably further enhanced the recovery of its scallop population.

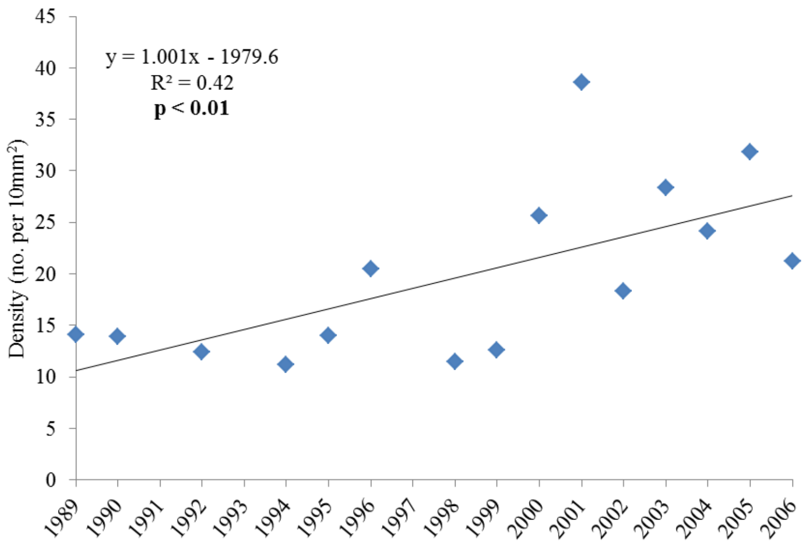

Figure 14 | Mean density of all mobile benthic invertebrate species (number of individuals per 100 m^2) in the Port Erin closed area on the Isle of Man between 1989 and 2006. The regression indicates a significant increase over time. Taken from Brown (2013).

Due to the success of the Port Erin closed area, both in terms of fisheries and conservation benefits, the Isle of Man government has subsequently established a network of similar protected areas around the island (Fig. 15). Importantly, the local fishing industry is now strongly supportive of these spatial management measures and is actively involved in related research, monitoring and stock enhancement exercises (Beukers-Stewart & Brand 2007).

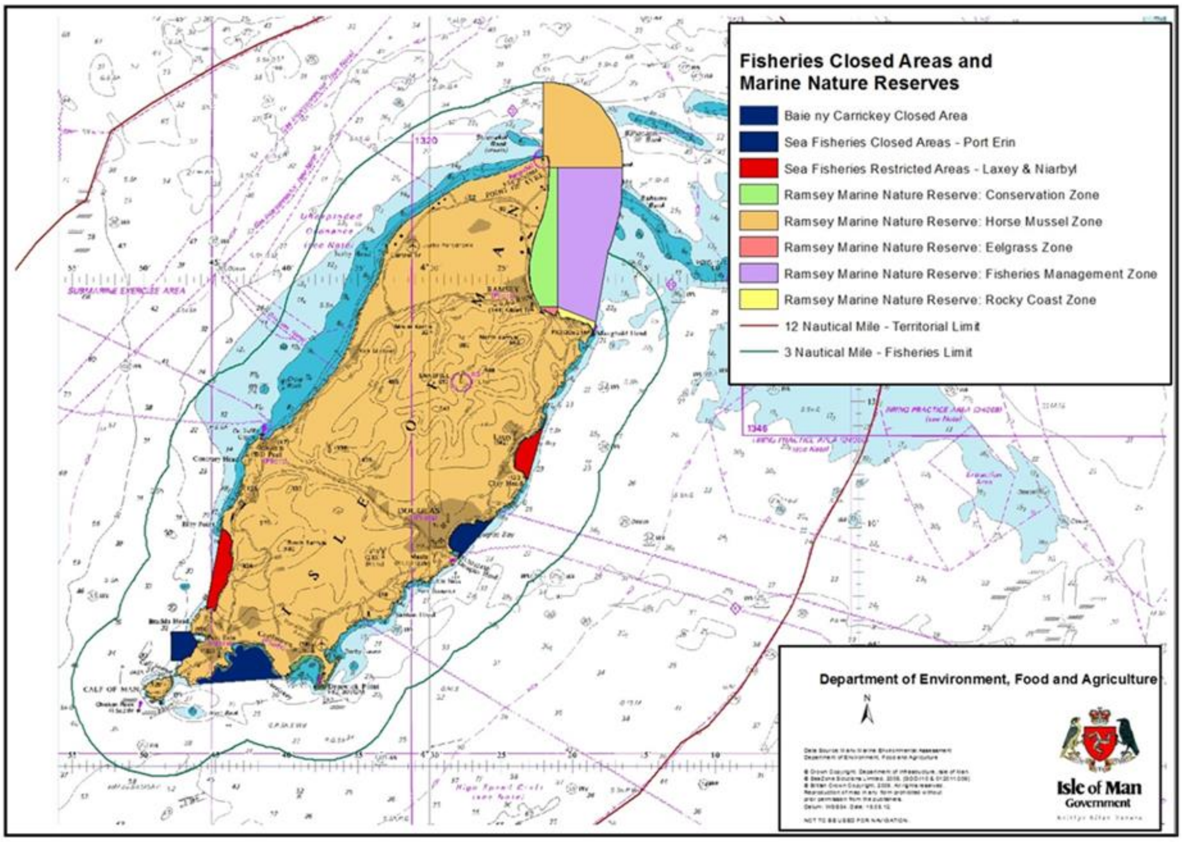

Figure 15 | Fisheries closed areas and marine nature reserves around the Isle of Man, as of November 2012 (Department of Environment, Food and Agriculture, Isle of Man government).

3.2.2. Lyme Bay Marine Protected Area (MPA), Dorset and Devon, England

Located on the south west coast of England, Lyme Bay is an area renowned for its rocky reefs formed of mudstone, limestone, chalk and granite outcrops. In addition to being listed under Annex I of the EU Habitats Directive, the reefs also support extremely high levels of biodiversity including important and structurally complex habitats such as ross coral, *Pentapora fascialis*, dead man's fingers, *Alcyonium digitaum*, and the iconic pink sea fan, *Eunicella verrucosa* (Fig. 16) which is listed under Schedule 5 of the UK Wildlife and Countryside Act 1981 (Sheehan et al. 2013b). In forming biogenic reefs these species provide important nursery habitats, offering many species protection from predation and surfaces for larval settlement (Bradshaw et al. 2001, 2003; Beck et al. 2001; Kamenos et al. 2004b, 2004a; Gibb et al. 2007; Laurel et al. 2009). Concerns over the impacts of towed fishing gears on the reefs resulted in the establishment of four small voluntary areas closed to towed gears (totalling 22 km^2) between 2001 and 2006. In 2008, lack of compliance with previous measures led to the closed areas being combined to form one large statutory MPA which excluded towed gears from an area of 206 km^2. Static gear fisheries, including potting and netting, were permitted to continue, along with SCUBA diving for scallops and sea angling (Sheehan et al. 2013b).

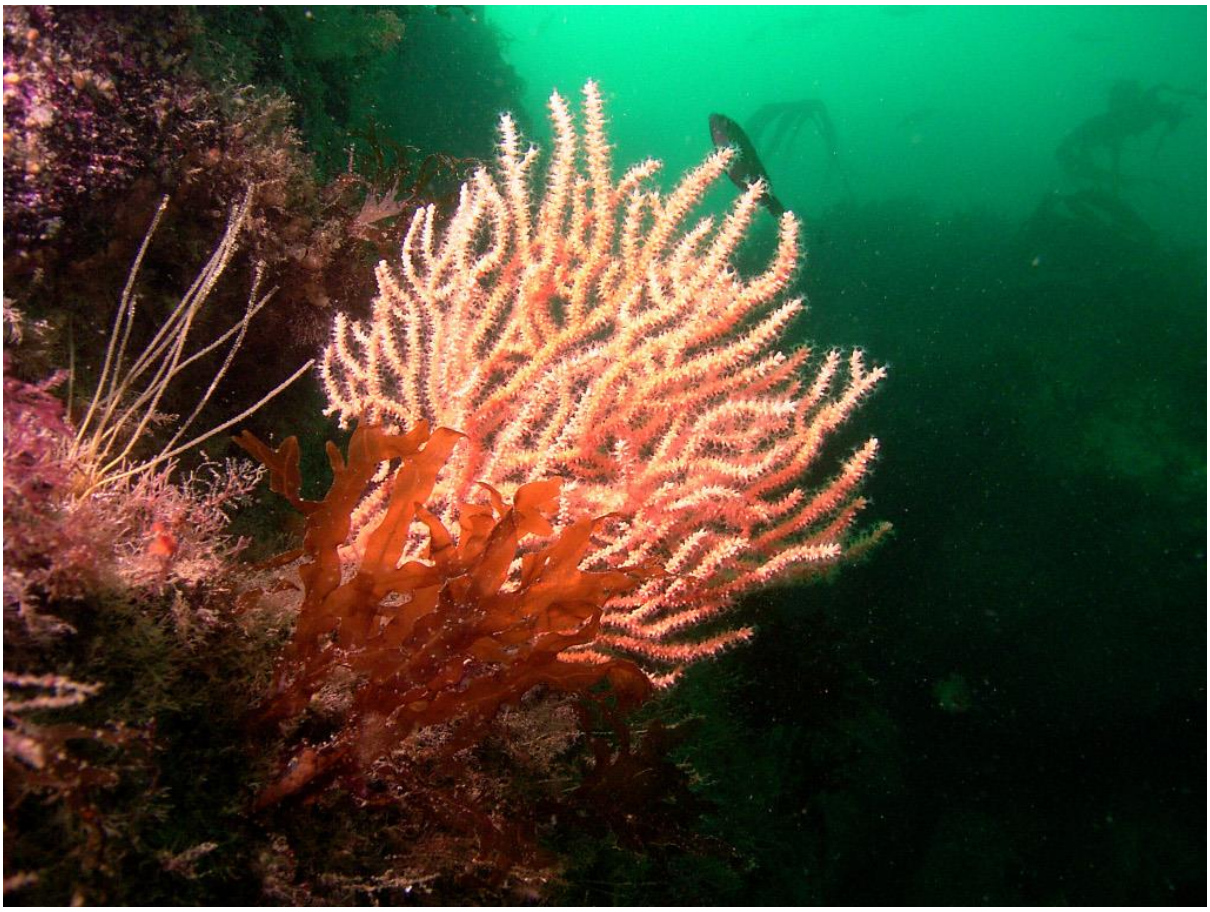

Figure 16 | The iconic pink sea fan is protected under the UK Wildlife and Countryside Act 1981 but is very vulnerable to the impacts of scallop dredging. Image taken from www.lusac.org.uk

Due to the scraping action of trawls and dredges operating in the area, boulders and cobbles inside the newly protected area had limited sessile and epifaunal life growing on them when monitoring first began (Sheehan et al. 2013b). However, observations made three years later revealed structural complexity had substantially increased within the MPA (Fig. 17) through the recovery of pink sea fans (increase of 636%), ross coral (increase of 385%), branched sponges (increase of 414%) and hydroids (increase of 229%; Sheehan et al. 2013a, b). Furthermore, survey data also revealed that these reef-associated species had also colonised sedimentary habitat adjacent to what was originally perceived as reef (Sheehan et al 2013a). This suggests the functional extent of reef was greater than its visual boundary. Such species are known to improve survivorship of juvenile fish by acting as important fishery nursery areas and feeding grounds (Auster et al. 1996; Bradshaw et al. 2001, 2003). In addition, the main target species of the excluded fishery, the commercially valuable king scallop, was also found to be in a state of recovery within the MPA (Sheehan et al. 2013b). Continued recovery of this species could provide fisheries benefits to surrounding open grounds through larval export (see above and below).

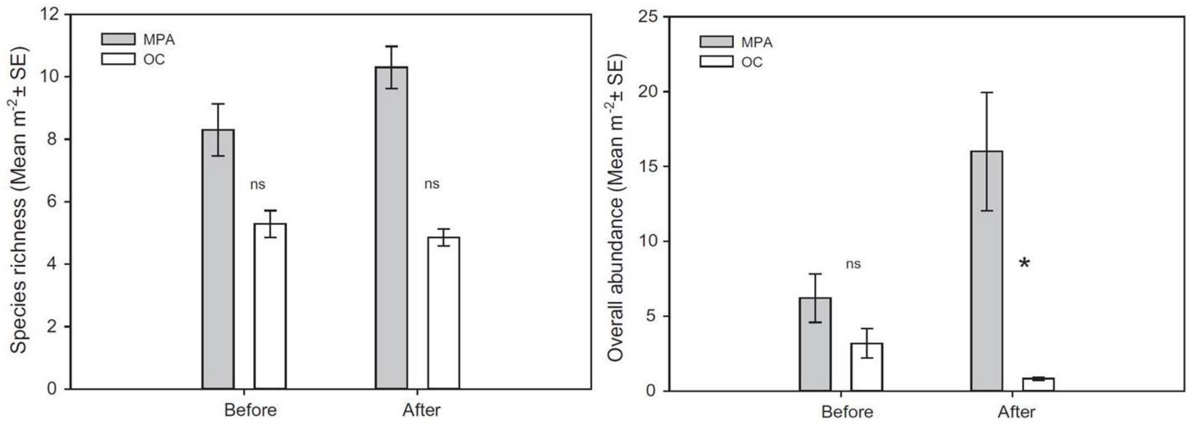

Figure 17 | After three years of protection, the mean species richness (left) and abundance (right) of reef associated species had substantially increased within Lyme Bay MPA. Lower axis represents 'Before' and 'After' 3 years of protection and between Treatments (MPA = Marine Protected Area; OC = Open Control). Taken from Sheehan et al. (2013a).

After the MPA was declared in 2008, a dramatic increase in potting effort occurred as static fishers were able to operate without disturbance from mobile gears. However, this lead to concerns about impacts of increased effort on the target species (mostly edible crab) and seabed habitats. An innovative new project, involving collaboration between scientists and fishermen, is now helping to determine sustainable levels of potting and to develop a management plan (www.lymebayreserve.co.uk/conservation-and-science/research).

3.2.3. Lamlash Bay No-Take Zone (NTZ), Isle of Arran, Scotland

In September 2008, a fully protected No-Take Zone (NTZ) measuring 2.67 km^2 in area was established in Lamlash Bay, Isle of Arran, Scotland, thereby prohibiting all sea fishing within the reserve under the Inshore Fishing (Scotland) Act of 1984 (Axelsson et al. 2009). The Firth of Clyde, in which the Isle of Arran sits, is regarded as one of the most degraded marine environments in the UK, primarily due to over a century of intensive fisheries exploitation (Thurstan & Roberts 2010; Howarth et al. 2013). The NTZ was therefore passed by the Scottish parliament under the rationale that the reduction in fishing pressure would help regenerate the local marine environment and enhance commercial shellfish and fish populations in and around Lamlash Bay. Lamlash Bay is the first and only fully protected NTZ in Scotland, and the only statutory marine reserve in the UK that was originally proposed and campaigned for by a local community group, The Community of Arran Seabed Trust (COAST; Prior 2011). Lamlash Bay is also unique in that the majority of MPAs in the UK were proposed either for conservation (e.g. Lundy Marine Nature reserve and Lyme Bay Marine Reserve) or fishery purposes (e.g. closed areas off the Isle of Man), not for both.

After four years of protection, important nursery habitats such as macroalgae and hydroids were twice as abundant within the NTZ compared to neighbouring fishing grounds, and their abundance has been steadily increasing over time (Howarth et al. In prep). In addition, the recovery of nursery habitats was found to result in higher levels of settlement by juvenile scallops (Howarth et al. 2011) meaning juvenile scallop abundance was more than 350% higher within the NTZ than outside in some years (Howarth et al. In prep). These results provide evidence that protecting areas from fishing can allow seafloor habitats to recover, thereby generating a number of benefits that flow back to species of commercial interest. In the long term, these effects will increase the numbers of juvenile scallops entering the adult stock as a greater proportion of juveniles survive to reach maturity (Beukers-Stewart et al. 2003; Vause et al. 2007).

When monitoring began in 2010 it appeared that, despite providing benefits to juvenile scallops, the Lamlash Bay NTZ was yet to have any effect on the density of adult scallops (Howarth et al. 2011). Since then, the density of king scallops has increased steadily, with legal sized scallops becoming 60% more abundant within the NTZ than outside by 2012 (Fig. 18). As scallops breed by releasing both male and female gametes into the water column during synchronised spawning events (Brand 2006b), the increase in population density will likely result in a rapid increase in fertilisation success (Macleod et al. 1985; Stoner & Ray-Culp 2000; Vause et al. 2007).

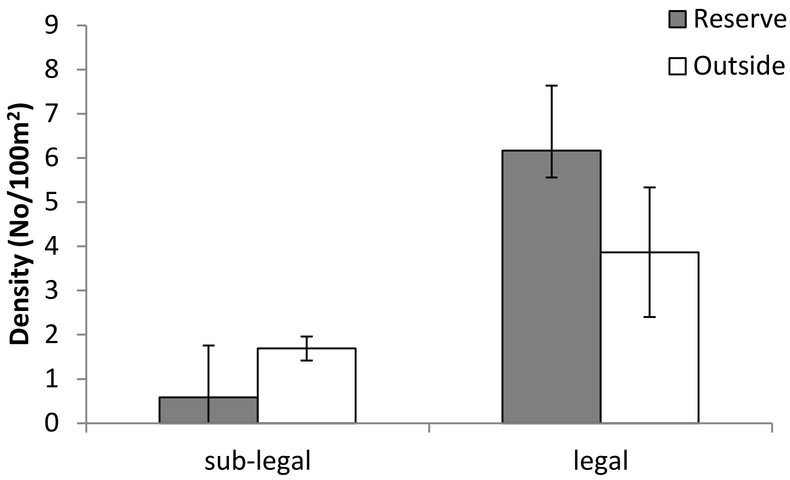

Figure 18 | Density (number of individuals per 100 m^2) of different size classes of king scallops sampled within and outside Lamlash Bay marine reserve after four years of protection in 2012. Error bars represent ±1 SE. Taken from Howarth et al. (In prep).

As well as increasing scallop density, evidence suggests that the NTZ is enabling the age and size structure of scallop populations within its boundaries to return to a more natural and extended state (Fig. 19). After four years of protection, it was found that king scallops were on average 25mm larger and 1.6 years older within the NTZ than outside (Howarth et al. In prep). Likewise, queen scallops were 12mm larger and 0.7 years older within the NTZ. As larger, older scallops produce considerably greater quantities of eggs per individual (Bradshaw et al. 2001; Beukers-Stewart et al. 2005) these effects should result in higher levels of reproduction and recruitment to surrounding fishing grounds (Beck et al. 2001; Gibb et al. 2007; Laurel et al. 2009). In further support of this, the reproductive biomass of king scallops per unit area was 185% greater within the NTZ than on surrounding fishing grounds by 2012 (Fig. 20).

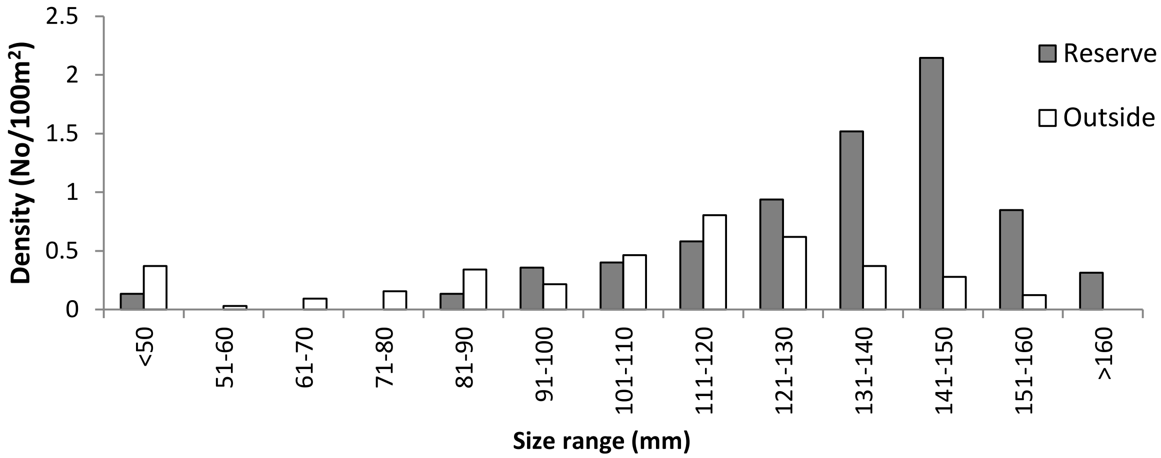

Figure 19 | Size structure (number of individuals per $100m^2$ per size category) of king scallop populations sampled within and outside Lamlash Bay marine reserve after four years of protection in 2012. The graph shows there are significantly more large-bodied scallops within the reserve. Taken from Howarth et al. (In prep).

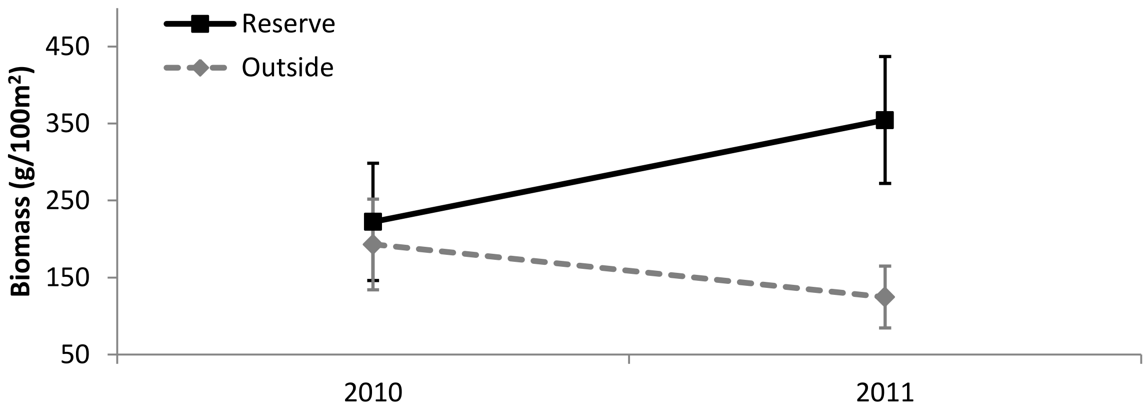

Figure 20 | Mean reproductive biomass (g per $100m^2$) of king scallops within and outside the NTZ for two years when scallop dissections were conducted. Error bars represent ±1 SE. Taken from Howarth et al. (In prep).

The NTZ in Lamlash Bay also appears to be generating fishery benefits for other commercially important species. After five years protection (by 2013), the Catch Per Unit Effort (CPUE) of legal sized European lobsters was 189% higher within the NTZ than outside (Howarth et al. In prep). Furthermore, the CPUE, weight and size of lobsters were all found to significantly decline with increasing distance from the NTZ (Fig. 21). This may be evidence that increasing lobster densities within the NTZ are increasing competition for space and resources, meaning lobsters are moving outside the boundaries of the NTZ to where densities are lower, and also to where they can contribute to fishery landings. Preliminary results from tagging of lobsters suggest they regularly move across the reserve boundaries. This recovery of lobsters within the NTZ may also be producing reproductive benefits. The potential number of eggs carried per female lobster (estimated from body size) was 27.3% higher within the NTZ. In addition, berried (egg-bearing) females were 5.5 times more abundant, suggesting that the 2.67 km^2 NTZ has a potential egg output equivalent to an unprotected area of 19.1 km^2. This further supports the idea that MPAs can contribute disproportionally to recruitment in relation to the actual size of area they protect (Beck et al. 2001; Gibb et al. 2007; Laurel et al. 2009; Harrison et al. 2012).

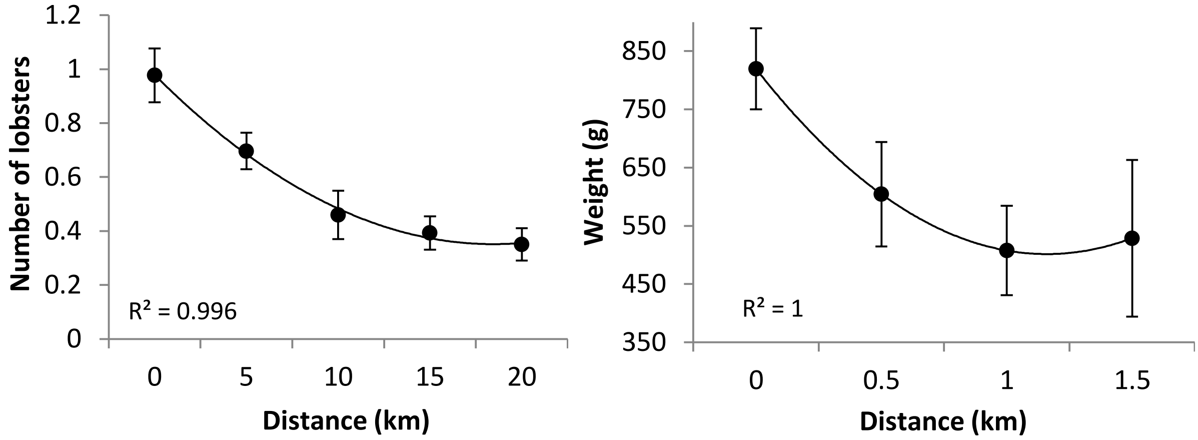

Figure 21 | Mean number of lobsters caught (left) and average weight of lobster caught (right) per pot for the years 2012 and 2013 combined, plotted against distance from the boundaries of Lamlash Bay NTZ. A distance of 0 represents sites located within the NTZ. The data have been fitted with a polynomial trend line and the resulting R^2 values are displayed. Error bars represent ±1 Standard Error (SE). Howarth et al. (In prep.).

Overall, evidence from Lamlash Bay suggests that protected areas can act as a safe haven for those individuals within their boundaries, allowing them to reach sexual maturity, greater fecundity, greater densities and larger sizes. This suggests that protected areas can be a useful tool in ecosystem-based fishery management and that, by providing fishery and ecological benefits, they can allow seafloor habitats to recover whilst safeguarding the long-term sustainability of commercially important shellfish stocks.

4. Conclusions and recommendations

Landings of king scallops are growing faster than any other commercially targeted species in the UK. Generating over £66.9 million per year, king scallops represent the UK's second most valuable fishery resource, over 95% of which are caught by scallop dredgers. Queen scallops, although more commonly targeted by trawlers than dredgers, also represent a substantial economic resource. Despite their growing importance, there is considerable evidence that the management of UK scallop fisheries could be significantly improved. This is because the fishery currently has a large number of negative impacts on marine ecosystems and commercial stocks. By damaging seafloor habitats, scallop dredging not only significantly reduces biodiversity; it also damages much of the habitat that is crucial for the settlement and survival of juvenile scallops, as well as a number of other species of commercial importance. A new management regime for UK scallop fisheries that provided better protection to vital scallop nursery and breeding areas would undoubtedly result in more productive and sustainable fisheries, and maintain healthier benthic ecosystems.

Excluding scallop dredging from selected areas of the seabed can resolve conflict between fisheries and generate ecological and fishery benefits by providing spawning refuges for the replenishment of stocks and allowing damaged seafloor habitats to regenerate. In the case studies presented, the benefits of excluding towed fishing gears have outweighed the costs of losing access to some fishing grounds by increasing biodiversity and recruitment to commercial fisheries. We therefore believe that a network of protected areas around the UK, both including and beyond what is currently in place and being proposed, would provide substantial benefits to the scallop fishery, and reduce its impact on the wider ecosystem. The following principles should be used to guide the development of this network:

- ***Protected areas should be strategically located and designed to offer multiple benefits wherever possible***. This includes maximising the potential for larval export / spillover of scallops and other commercial species, offering protection to biodiverse and vulnerable areas and reducing conflict between static and mobile fisheries.
- ***Scallop dredging should be excluded from vulnerable habitats within existing and future protected areas at the site level, rather than just specifically where vulnerable features currently exist.*** Experience from the above case studies demonstrates both the need for buffer zones around vulnerable features, and that recovery of such habitats can extend beyond what was originally perceived to be likely.
- ***Protected areas should not just cover the most vulnerable habitat types, but ensure representation of the full of range of substrates and biodiversity***. All habitat types contribute to biodiversity and fisheries in different ways and therefore should be afforded some element of protection.
- ***Protected areas should be permanent to maximise benefits to fisheries and conservation***. The above case studies of protected areas indicate that recovery of both scallops and other benthic species has been less rapid in the UK than in some other areas in the world. The use of rotational closed areas would therefore not be appropriate here as the benefits gained from protection would be quickly lost but slow to re-gain.

- ***Protected areas should be well monitored in order to assess performance***. This will inform management strategies and be crucial for communication with stakeholders. Given the uncertain, but likely substantial effects of ocean warming and acidification in the future, some adjustments to the protected areas are likely to be necessary over time.

Closing some areas to fishing may have some short term negative effects on local economies and the welfare of coastal communities. If these short term costs can be overcome, the scallop fishing industry is one of the economic groups with the most to gain in the long term. However, the same industry also has the most potential to impact on the success of this approach (Pita et al. 2013). Fishers must therefore be actively involved in the decision making process when closed areas are being established and emphasis should be placed on the fishery benefits that closed areas can afford. Spatial management of the UK scallop dredge fishery, as described above, will go a long way towards ensuring it has a sustainable and productive future while reducing its impact on the wider ecosystem. However, there is also an urgent need to reduce current overall effort in the fishery to sustainable levels and to develop more environmentally friendly scallop dredges. Furthermore, we would promote the development of local management of scallop fisheries, particularly in the inshore sector, to encourage enhanced levels of stewardship within the industry.

References

Airoldi, L., D. Balata, and M. W. Beck. 2008. The Gray Zone: Relationships between habitat loss and marine diversity and their applications in conservation. Journal of Experimental Marine Biology and Ecology 366:8–15.

Andersen, S., E. S. Grefsrud, and T. Harboe. 2013. Effect of increased pCO2 level on early shell development in great scallop (*Pecten maximus,* Lamarck) larvae. Biogeosciences 10:6161–6184.

Auster, P. J., R. J. Malatesta, R. W. Langton, L. Watting, P. C. Valentine, C. L. S. Donaldson, E. W. Langton, A. N. Shepard, and W. G. Babb. 1996. The impacts of mobile fishing gear on seafloor habitats in the gulf of Maine (Northwest Atlantic): Implications for conservation of fish populations. Reviews in Fisheries Science 4:185–202.

Axelsson, M., S. Dewey, L. Plastow, and J. Doran. 2009. Mapping of marine habitats and species within the Community Marine Conservation Area at Lamlash Bay. Page 130. Aberdeen.

Barreto, E., and N. Bailey. 2013. Fish and shellfish stocks. 2013 edition. 65 pages. Edinburgh.

BBC. 2012. Scallop dredging ban wanted by East Yorkshire fishermen. Accessed February 11th 2014 from http://www.bbc.co.uk/news/uk-england-humber-17861822.

Beck, M. W. et al. 2001. The Identification, Conservation, and Management of Estuarine and Marine Nurseries for Fish and Invertebrates. BioScience 51:633–641. American Institute of Biological Sciences.

Beukers-Stewart, B. D., and Brand, A. R. 2007. Seeking sustainable scallops: do MPAs really work? Coastal Futures Conference. January 2007, London, UK.

Beukers-Stewart, B. D., and J. S. Beukers-Stewart. 2009. Principles for the Management of Inshore Scallop Fisheries around the United Kingdom. University of York.

Beukers-Stewart, B. D., S. R. Jenkins, and A. R. Brand, 2001. The efficiency and selectivity of spring-toothed scallop dredges: a comparison of direct and indirect methods of assessment. Journal of Shellfish Research 20:121–126.

Beukers-Stewart, B. D., W. Lart, C. Sinfield, A. R. Brand, and S. R. Jenkins. 2012. Patterns and mechanisms of damage in a scallop dredge fishery. World Fisheries Congress, May 2012, Edinburgh, Scotland, UK.

Beukers-Stewart, B. D., M. W. J. Mosley, and A. R. Brand. 2003. Population dynamics and predictions in the Isle of Man fishery for the great scallop (*Pecten maximus* L.). ICES Journal of Marine Science 3139:224–242.

Beukers-Stewart, B. D., Vause, B. J., Mosley, M. W., and A.R. Brand. 2004. Evidence for larval export of scallops from a small closed area off the Isle of Man. ICES Annual Science Conference, September 2004, Vigo, Spain.

Beukers-Stewart, B. D., B. J. Vause, M. W. J. Mosley, H. Rossetti, and A. R. Brand. 2005. Benefits of closed area protection for a population of scallops. Marine Ecology Progress Series 298:189–204.

Birkett, D. A., C. Maggs, and M. J. Dring. 1998. Maerl. An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project). Page 116.

Blyth, R. E., M. J. Kaiser, G. Edwards-Jones, and P. J. B. Hart. 2002. Voluntary management in an inshore fishery has conservation benefits. Environmental Conservation 29:493–508.

Blyth, R., M. Kaiser, G. Edwards-Jones, and P. Hart. 2004. Implications of a zoned fishery management system for marine benthic communities. Journal of Applied Ecology 41:951–961.

Blyth-Skyrme, R., M. Kaiser, J. Hiddink, G. Edwards-Jones, and P. Hart. 2006. Conservation Benefits of Temperate Marine Protected Areas: Variation among Fish Species. Conservation Biology 20:811–820.

Boersma, P. D., and J. K. Parrish. 1999. Limiting abuse: marine protected areas, a limited solution. Ecological Economics 31:287–304.

Boulcott, P., and T. R. W. Howell. 2011. The impact of scallop dredging on rocky-reef substrata. Fisheries Research 110:415–420.

Boulcott, P., C. P. Millar, and R. J. Fryer. 2014. Impact of scallop dredging on benthic epifauna in a mixed-substrate habitat. ICES Journal of Marine Science, doi.10.1093/icesjms/fst197.

Bradshaw, C., P. Collins, and A. Brand. 2003. To what extent does upright sessile epifauna affect benthic biodiversity and community composition? Marine Biology 143:783–791.

Bradshaw, C., L. Veale, A. Hill, and A. R. Brand. 2000. The effects of scallop dredging on gravelly sea-bed communities. Pages 83–104 in M. Kaiser and S. de Groot, editors. Effects of fishing on non-target species and habitats. Gray Publishing, Tunbridge Wells.

Bradshaw, C., L. Veale, A. Hill, and A. R. Brand. 2001. The effect of scallop dredging on Irish Sea benthos: experiments using a closed area. Hydrobiologia 465:129–138.

Bradshaw, C., L. O. Veale, A. S. Hill, and A. R. Brand. 2002. The effect of scallop dredging on Irish Sea benthos: experiments using a closed area. Pages 129–138 in G. Burnell and H. J. Dumont, editors. Coastal Shellfish — A Sustainable Resource. Springer Netherlands.

Brand, A. R. 2006a. The European Scallop Fisheries for *Pecten maximus*, *Aequipecten opercularis* and *Mimachlamys varia*. Pages 991-1058 in S. Shumway and G. Parsons, editors. Scallops: biology, ecology and aquaculture. Elsevier, Amsterdam.

Brand, A. R. 2006b. Scallop Ecology: Distributions and Behaviour. Pages 651-744 in S. Shumway and G. Parson, editors. Scallops: biology, ecology and aquaculture. Elsevier, Amsterdam.

Brand, A. R., J. D. Paul, and J. N. Hoogestegger. 1980. Spat settlement of the scallops *Chlamys opercularis* (L.) and *Pecten maximus* (L.) on artificial collectors. Journal of the Marine Biological Association of the UK 60:379–390.

Brown, R. L. 2013. Untangling the effects of fishing effort and environmental variables on benthic communities of commercially fished scallop grounds. PhD thesis. University of York.

Cacabelos, E., C. Olabarria, M. Incera, and J. Troncoso. 2010. Effects of habitat structure and tidal height on epifaunal assemblages associated with macroalgae. Estuarine, Coastal and Shelf Science 89:43–52.

Callier, M., C. McKindsery, P. Archambault, and G. Desgrosiers. 2009. Responses of benthic macrofauna and biogeochemical fluxes to various levels of mussel biodeposition: an in situ "benthocosm" experiment. Marine Pollution Bulletin 58:1544–1553.

Capell, R., M. Robinson, J. Gascoigne, and F. Nimmo. 2013. A review of the Scottish scallop review. Poseidon report to Marine Scotland, December 2013. 82 pp.

Christie, H., N. Jørgensen, and K. Norderhaug. 2007. Bushy or smooth, high or low; importance of habitat architecture and vertical position for distribution of fauna on kelp. Journal of Sea Research 58:198–208.

Collie, J. S., G. A. Escanero, and P. C. Valentine. 1997. Effects of bottom fishing on the benthic megafauna of Georges Bank. Marine Ecology Progress Series 155:159–172.

Collie, J. S., J. M. Hall-Spencer, M. J. Kaiser, and I. R. Poiner. 2000. A quantitative analysis of fishing impacts on shelf-sea benthos. Journal of Animal Ecology 69:785–798.

Consultation on the Recommendations from the Scottish Licensing Review Working Group. 2013. Page 32. Retrieved from http://www.scotland.gov.uk/Publications/2013/12/1295/0.

Cook, R. et al. 2013. The substantial first impact of bottom fishing on rare biodiversity hotspots: a dilemma for evidence-based conservation. PloS one 8:e69904.

Craven, H. R., A. R. Brand, and B. D. Stewart. 2013. Patterns and impacts of fish bycatch in a scallop dredge fishery. Aquatic Conservation: Marine and Freshwater Ecosystems 23:152–170.

Currie, D., and G. Parry. 1996. Effects of scallop dredging on a soft sediment community: a large-scale experimental study. Marine Ecology Progress Series 134:131–150.

Daan, N., P. Bromley, J. Hislop, and N. Nielsen. 1990. Ecology of North Sea fish. Netherlands Journal of Sea Research 26:343–386.

Dale, A., P. Boulcott, and T. Sherwin. 2011. Sedimentation patterns caused by scallop dredging in a physically dynamic environment. Marine Pollution Bulletin 62:2433–2441.

Dare, P., D. Key, and P. M. Connor. 1993. The efficiency of spring-loaded dredges used in the Western English channel fishery for scallops, *Pecten maximus* (L.). ICES Report No. CM 1993/B:15. Copenhagen, Denmark.

Dayton, P. K., S. F. Thrush, M. T. Agardy, and R. J. Hofman. 1995. Environmental effects of marine fishing. Aquatic Conservation: Marine and Freshwater Ecosystems 5:205–232.

DEFA. 2013. Proposals for the Management of the Isle of Man Queen Scallop Fishery 2014. Page 14. Retrieved from https://www.gov.im/lib/docs/daff/Consultations/2014qscconsultationfinal.pdf.

DEFRA. 2013a. Towed gear: conservation rules. Page 9. Retrieved from https://www.gov.uk/government/publications/towed-gear-conservation-rules-part-3.

DEFRA. 2013b. Fishing Focus. The DEFRA and MMO marine fisheries newsletter. Page 8. Retrieved from https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/277499/fishing-focus-32-winter2013.pdf.

Dernie, K., M. Kaiser, E. Richardson, and R. Warwick. 2003. Recovery of soft sediment communities and habitats following physical disturbance. Journal of Experimental Marine Biology and Ecology 285-286:415–434.

Dobby, H., S. Millar, L. Blackadder, J. Turriff, and A. McLay. 2012. Scottish Scallop Stocks: Results of 2011 Stock Assessments. Marine Science Scotland, Edinburgh. 158pp.

DOE. 2005. Northern Ireland Habitat Action Plan *Modiolus modiolus* beds. Department of the Environment. Northern Ireland. Page 13.

Dolmer, P., and E. Stenalt. 2010. The impact of the adult blue mussel (*Mytilus edulis*) population on settling of conspecific larvae. Aquaculture International 18:3–17.

Doney, S. C., V. J. Fabry, R. A. Feely, and J. A. Kleypas. 2009. Ocean Acidification: The Other CO2 Problem. Annual Review of Marine Science 1:169–192.

Eleftheriou, A., and M. Robertson. 1992. The effects of experimental scallop dredging on the fauna and physical environment of a shallow sandy community. Netherlands Journal of Sea Research 299:289–299.

Enever, R., A. Revill, and A. Grant. 2007. Discarding in the English Channel, Western Approaches, Celtic and Irish seas (ICES subarea VII). Fisheries Research 86:143–152.

Fenberg, P. B. et al. 2012. The science of European marine reserves: Status, efficacy, and future needs. Marine Policy 36:1012–1021.

Foster, M. 2001. Rhodoliths: between rocks and soft places. Journal of Phycology 37:659–667.

Frank, K. T., and D. Brickman. 2001. Contemporary management issues confronting fisheries science. Journal of Sea Research 45:173–187.

Gaines, S. D., B. Gaylord, and J. L. Largier. 2003. Avoiding current oversights in marine reserve design. Ecological Applications 12:S32–S46.

Gazeau, F., C. Quiblier, J. M. Jansen, J.-P. Gattuso, J. J. Middelburg, and C. H. R. Heip. 2007. Impact of elevated CO2 on shellfish calcification. Geophysical Research Letters 34:L07603.

Gibb, F. M., I. M. Gibb, and P. J. Wright. 2007. Isolation of Atlantic cod (*Gadus morhua*) nursery areas. Marine Biology 151:1185–1194.

Giraud, G., and J. Cabioch. 1976. Ultrastructural study of the activity of superficial cells of the thallus of the Corallinaceae (Rhodophyceae). Phycologia 13:405–414.

Grall, J., and J. Hall-Spencer. 2003. Problems facing maerl conservation in Brittany. Aquatic Conservation: Marine and Freshwater Ecosystems: S55–S64.

Grantham, B. A., G. L. Eckert, and A. L. Shanks. 2003. Dispersal potential of marine invertebrates in diverse habitats. Ecological Applications 13:S108–S116.

Hall-Spencer, J., J. Kelly, and C. Maggs. 2008. Assessment of maerl beds in the OSPAR area and the development of a monitoring program. Report for the Department of the Environment, Heritage & Local Government of Ireland. Page 34.

Hall-Spencer, J., and P. Moore. 2000. Scallop dredging has profound, long-term impacts on maerl habitats. ICES Journal of Marine Science 57:1407–1415.

Hall-Spencer, J., N. White, E. Gillespie, and A. Foggo. 2006. Impact of fish farms on maerl beds in strongly tidal areas. Marine Ecology Progress Series 326:1–9.

Halpern, B. S. 2003. The impact of marine reserves: do reserves work and does reserve size matter? Ecological Applications 13:117–137. Ecological Society of America.

Halpern, B. S., and R. R. Warner. 2002. Marine reserves have rapid and lasting effects. Ecology Letters 5:361–366.

Hargrave, B., L. Doucette, P. Cranford, B. Law, and T. Milligan. 2008. Influence of mussel aquaculture on sediment organic enrichment in a nutrient-rich coastal embayment. Marine Ecology Progress Series 365:137–149.

Harrison, H. B., D. H. Williamson, R. D. Evans, G. R. Almany, S. R. Thorrold, G. R. Russ, K. A. Feldheim, L. van Herwerden, S. Planes, M. Srinivasan, M. L. Berumen and G. P. Jones (2012) Larval export from marine reserves and the recruitment benefit for fish and fisheries. Current Biology 22, 1023–1028.

Hart, P. J. B. 1998. Enlarging the shadow of the future: avoiding conflict and conserving fish. Fish and Fisheries Series 23:227–238.

Hinz, H., L. G. Murray, F. R. Malcolm, and M. J. Kaiser. 2012. The environmental impacts of three different queen scallop (*Aequipecten opercularis*) fishing gears. Marine environmental research 73:85–95.

Hinz, H., D. Tarrant, a Ridgeway, M. Kaiser, and J. Hiddink. 2011. Effects of scallop dredging on temperate reef fauna. Marine Ecology Progress Series 432:91–102.

Howarth, L. M., C. M. Roberts, R. H. Thurstan, and B. D. Stewart. 2013. The unintended consequences of simplifying the sea: making the case for complexity. Fish and Fisheries DOI: 10.1111/faf.12041.

Howarth, L. M., H. L. Wood, A. P. Turner, and B. D. Beukers-Stewart. 2011. Complex habitat boosts scallop recruitment in a fully protected marine reserve. Marine Biology 158:1767–1780.

Howell, T. R. W., S. E. B. Davis, J. Donald, H. Dobby, I. Tuck, and N. Bailey. 2006. Report of marine laboratory scallop stock assessments. Page 152. Aberdeen.

Howell, T. R. W., and D. I. Fraser. 1984. Observations on the dispersal and mortality of the scallop *Pecten maximus* (L.). ICES Council Shellfish Committee, CM 1984/K35. ICES Journal of Marine Science. Copehagen, Denmark.

Hsieh, C.-H., C. S. Reiss, J. R. Hunter, J. R. Beddington, R. M. May, and G. Sugihara. 2006. Fishing elevates variability in the abundance of exploited species. Nature 443:859–62.

ICES. 1992. Report of the ices working grow on the effects of extraction of marine sediments on fisheries. ICES cooperative research report No. 182. Page 84. Copehagen, Denmark.

ICES. 2001. Effects of Extraction of Marine Sediments on the Marine Ecosystem. ICES Cooperative Research Report No. 247. Page 84. Copehagen, Denmark.

Jenkins, S. R., B. D. Beukers-Stewart, and A. R. Brand. 2001. Impact of scallop dredging on benthic megafauna: a comparison of damage levels in captured and non-captured organisms. Marine Ecology Progress Series 215:297–301.

Jenkins, S. R., and A. R. Brand. 2001. The effect of dredge capture on the escape response of the great scallop, *Pecten maximus* (L.) : implications for the survival of undersized discards. Journal of Experimental Marine Biology and Ecology 266:33–50.

Jenkins, S. R., W. Lart, B. J. Vause, and A. R. Brand. 2003. Seasonal swimming behaviour in the queen scallop (*Aequipecten opercularis*) and its effect on dredge fisheries. Journal of Experimental Marine Biology and Ecology 289:163–179.

Jenkins, S. R., C. Mullen, and A. R. Brand. 2004. Predator and scavenger aggregation to discarded by-catch from dredge fisheries: importance of damage level. Journal of Sea Research 51:69–76.

Jennings, S., and M. J. Kaiser. 1998. The Effects of Fishing on Marine Ecosystems. Advances in Marine Biology 34:201–352.

JNCC. 2013. UK Protected sites. Accessed May 20th 2013, from http://jncc.defra.gov.uk/default.aspx?page=4.

Jones, P. J. S. 2007. Point-of-View: Arguments for conventional fisheries management and against no-take marine protected areas: only half of the story? Reviews in Fish Biology and Fisheries 17:31–43.

Jones, P. J. S. 2012. Marine protected areas in the UK: challenges in combining top-down and bottom-up approaches to governance. Environmental Conservation 39:248–258.

Kaiser, M. J. et al. 1996. Benthic disturbance by fishing gear in the Irish Sea: a comparison of beam trawling and scallop dredging. Aquatic Conservation: Marine and Freshwater Ecosystems 6:269–285.

Kaiser, M. J. 2005. Are marine protected areas a red herring or fisheries panacea? Canadian Journal of Fisheries and Aquatic Sciences 62:1194–1199.

Kaiser, M. J. 2007a. Just how "green" is diving for scallops? Fishing News. London.

Kaiser, M. J. 2007b. A summary of the impacts of scallop dredging on seabed biota and habitats. Report for Natural England.

Kaiser, M. J., M. Attrill, S. Jennings, D. Thomas, D. Barnes, A. Brierly, N. Polunin, D. Rafaelli, and P. Williams. 2005. Marine Ecology. Processes, Systems and Impacts. Oxford University Press, Oxford.

Kaiser, M. J., K. Cheney, F. Spence, D. Edwards, and K. Radford. 1999. Fishing effects in northeast Atlantic shelf seas: patterns in fishing effort, diversity and community structure VII. The effects of trawling disturbance on the fauna associated with the tubeheads of serpulid worms. Fisheries Research 40:195–205.

Kaiser, M. J., K. Clarke, H. Hinz, M. Austen, P. Somerfield, and I. Karakassis. 2006. Global analysis of response and recovery of benthic biota to fishing. Marine Ecology Progress Series 311:1–14.

Kaiser, M. J., J. Collie, J. Hall-Spencer, S. Jennings, and I. Poiner. 2002. Modification of marine habitats by trawling activities: prognosis and solutions. Fish and Fisheries 3:114–136.

Kaiser, M. J. 2004. Marine protected areas: the importance of being earnest. Aquatic Conservation: Marine and Freshwater Ecosystems 14:635–638.

Kaiser, M. J., R. E. Blyth-Skyrme, P. J. B. Hart, G. Edwards-jones, and D. Palmer. 2007. Evidence for greater reproductive output per unit area in areas protected from fishing. Canadian Journal of Fisheries and Aquatic Sciences 64:1284–1289.

Kaiser, M. J., F. E. Spence, and P. J. B. Hart. 2000. Fishing gear restrictions and conservation of benthic habitat complexity. Conservation Biology 14:1512–1525

Kamenos, N. A., P. G. Moore, and J. M. Hall-Spencer. 2004a. Maerl grounds provide both refuge and high growth potential for juvenile queen scallops (*Aequipecten opercularis* L.). Journal of Experimental Marine Biology and Ecology 313:241–254.

Kamenos, N., P. G. Moore, and J. M. Hall-Spencer. 2004b. Nursery-area function of maerl grounds for juvenile queen scallops *Aequipecten opercularis* and other invertebrates. Marine Ecology Progress Series 274:183–189.

Kamenos, N., P. Moore, and J. Hall-Spencer. 2003. Substratum heterogeneity of dredged vs un-dredged maerl. Journal of the Marine Biological Association of the UK 83:411–413.

Kurihara, H. 2008. Effects of CO2-driven ocean acidification on the early developmental stages of invertebrates. Marine Ecology Progress Series 373:275–284.

Lambert, G., S. Jennings, M. Kaiser, H. Hinz, and J. Hiddink. 2011. Quantification and prediction of the impact of fishing on epifaunal communities. Marine Ecology Progress Series 430:71–86

Laurel, B. J., C. H. Ryer, B. Knoth, and A. W. Stoner. 2009. Temporal and ontogenetic shifts in habitat use of juvenile Pacific cod (*Gadus macrocephalus*). Journal of Experimental Marine Biology and Ecology 377:28–35.

LePennec, M., A. Paugam, and G. LePennec. 2003. The pelagic life of the pectinid *Pecten maximus*—review. ICES Journal of Marine Science 60:211–223.

Leslie, B., and R. Shelmerdine. 2007. Scallop Fishing in the Firth of Lorn Marine SAC: Review of Scientific Literature. SNH Contract no 19099.

Løkkeborg, S. 2005. Impacts of trawling and scallop dredging on benthic habitats and communities. FAO fisheries technical paper 472. Rome.

Macleod, J. A. A., J. P. Thorpe, and N. A. Duggan. 1985. A biochemical genetic study of population structure in queen scallop (Chlamys opercularis) stocks in the Northern Irish Sea. Marine Biology 87:77–82.

Marino II, M.C., Juanes, F., and K.D.E. Stokesbury. 2007. Effect of closed areas on populations of sea star *Asterias* spp. on Georges Bank. Marine Ecology Progress Series 347:29-49

McClanahan, T. R., and S. Mangi. 2000. Spillover of exploitable fishes from a marine park and its effect on the adjacent fishery. Ecological Applications 10:1792–1805. Ecological Society of America.

Metcalfe, K., T. Roberts, R. J. Smith, and S. R. Harrop. 2013. Marine conservation science and governance in North–West Europe: conservation planning and international law and policy. Marine Policy 39:289–295. Elsevier.

MMO. 2012. Evaluating the distribution, trends and value of inshore and offshore fisheries in England.

MMO. 2014. Revised approach to management of commercial fisheries in European marine sites in England. Retrieved from www.marinemanagement.org.uk/protecting/conservation/ems_fisheries.htm.

Monteiro, S., M. Chapman, and A. Underwood. 2002. Patches of the ascidian *Pyura stolonifera* (Heller, 1878): structure of habitat and associated intertidal assemblages. Journal of Experimental Marine Biology and Ecology 270:171–189.

Murray, L. G. 2013. The Isle of Man *Aequipecten opercularis* fishery stock assessment 2013. Bangor University Fisheries and Conservation Report No. 25. Pages 1–23.

Myers, R. A., S. D. Fuller, and D. G. Kehler. 2000. A fisheries management strategy robust to ignorance: rotational harvest in the presence of indirect fishing mortality. Canadian Journal of Fisheries and Aquatic Sciences 57:2357–2362.

Neill, S. P., and Kaiser, M. J. 2008. Sources and sinks of scallops (Pecten maximus) in the waters of the Isle of Man as predicted from particle tracking models. Fisheries & Conservation report No. 3, Bangor University.

Newell, R., L. Seiderer, and D. Hitchcock. 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. Oceanography and Marine Biology 36:127–178.

O'Neill, F., M. Robertson, K. Summerbell, M. Breen, and L. Robinson. 2013. The mobilisation of sediment and benthic infauna by scallop dredges. Marine Environmental Research 90:104–12.

Orensanz, J. M., A. M. Parma, and O. O. Iribarne. 1991. Population dynamics and management of natural stocks. Pages 625–713 in S. E. Shumway, editor. Scallops: biology, ecology and aquaculture. Elsevier, Amsterdam.

Palmer, D. 2006. The scallop fishery in England and Wales. Meeting of the North Western Waters Regional Advisory Committee, London, UK.

Paulet, Y. M., A. Lucas, and A. Gerard. 1988. Reproduction and larval development in two *Pecten maximus* (L.) populations from Brittany. Journal of Experimental Marine Biology and Ecology 119:145–156.

Pelc, R. A., R. R. Warner, S. D. Gaines, and C. B. Paris. 2010. Detecting larval export from marine reserves. Proceedings of the National Academy of Sciences 107:18266–18271.

Pikitch, E. K. et al. 2004. Ecosystem-Based Fishery Management. Science 305:346–347.

Pita, C., I. Theodossiou, and G. J. Pierce. 2013. The perceptions of Scottish inshore fishers about marine protected areas. Marine Policy 37:254–263.

Pomeroy, R. S., L. M. Watson, J. E. Parks, and G. a. Cid. 2005. How is your MPA doing? A methodology for evaluating the management effectiveness of marine protected areas. Ocean & Coastal Management 48:485–502.

Prior, S. 2011. Investigating the use of voluntary marine management in the protection of UK marine biodiversity. A Report for the Wales Environment Link (WEL) Marine Working Group. Page 46. Cardiff.

Probert, P. K. 1984. Disturbance, sediment stability, and trophic structure of soft-bottom communities. Journal of Marine Research 42:893–921.

Radford, L. 2013. UK sea fisheries statistics 2012. Page 182. London.

Ragnarsson, S. A., and J. M. Burgos. 2012. Separating the effects of a habitat modifier, *Modiolus modiolus* and substrate properties on the associated megafauna. Journal of Sea Research 72:55–63. Elsevier B.V.

Ragnarsson, S. A., and D. Raffaelli. 1999. Effects of the mussel *Mytilus edulis* L. on the invertebrate fauna of sediments. Journal of Experimental Marine Biology and Ecology 241:31–43.

Ramsay, K., and M. J. Kaiser. 1998. Demersal fishing disturbance increases predation risk for whelks (*Buccinum undatum* L.). Journal of Sea Research 39:299–304.

Rees, I. 2009. Background Document for *Modiolus modiolus* beds. Biodiversity Series for the OSPAR Commission. Page 30.

Rees, S. E., E. V. Sheehan, E. L. Jackson, S. C. Gall, S. L. Cousens, J-L. Solandt, M. Boyer, M. J. Attrill. 2013. A legal and ecological perspective of 'site integrity' to inform policy development and management of Special Areas of Conservation in Europe. Marine Pollution Bulletin 72: 14–21

Roberts, C. M., J. A. Bohnsack, F. R. Gell, J. P. Hawkins, and R. Goodridge. 2001. Effects of marine reserves on adjacent fisheries. Science 294:1920–1923.

Roberts, C. M., J. P. Hawkins, and F. R. Gell. 2005. The role of marine reserves in achieving sustainable fisheries. Philosophical Transactions of the Royal Society B: Biological Sciences 360:123–132.

Rodríquez-Cabello, C., A. Fernández, I. Olasao, and F. Sanchez. 2005. 2005. Survival of small-spotted catshark (*Scyliorhinus canicula*) discarded by trawlers in the Cantabrian Sea. Journal of the Marine Biological Association of the UK: 1145–1150.

Ryer, C., A. Stoner, and R. Titgen. 2004. Behavioural mechanisms underlying the refuge value of benthic habitat structure for two flatfishes with differing antipredator strategies. Marine Ecology Progress Series 268:231–243.

Sabine, C. L. et al. 2004. The Oceanic Sink for Anthropogenic CO_2. Science 305:367–371.

Sanderson, W., R. Holt, K. Ramsay, J. Perrins, A. McMath, and E. Rees. 2008. Small-scale variation within a *Modiolus modiolus* (Mollusca: Bivalvia) reef in the Irish Sea. II. Epifauna recorded by divers and cameras. Journal of the Marine Biological Association of the United Kingdom 88:143–149.

Sciberras, M., H. Hinz, J. Bennell, S. R. Jenkins, S. Hawkins, and M. J. Kaiser. 2013. Benthic community response to a scallop dredging closure within a dynamic seabed habitat. Marine Ecology Progress Series 480:83–98.

Scottish Government. 2014a. Policy. Protecting and sustainably using the marine environment. Supporting detail: Marine protected areas. Accessed 9th February 2014 from: www.gov.uk/government/policies/protecting-and-sustainably-using-the-marine-environment/supporting-pages/marine-protected-areas

Scottish Government. 2014b. Marine protected areas. Accessed 9th February 2014 from: http://www.scotland.gov.uk/Topics/marine/marine-environment/mpanetwork

Sheehan, E. V, S. L. Cousens, S. J. Nancollas, C. Stauss, J. Royle, and M. J. Attrill. 2013a. Drawing lines at the sand: evidence for functional vs. visual reef boundaries in temperate Marine Protected Areas. Marine pollution bulletin 76:194–202.

Sheehan, E. V, T. F. Stevens, S. C. Gall, S. L. Cousens, and M. J. Attrill. 2013b. Recovery of a Temperate Reef Assemblage in a Marine Protected Area following the Exclusion of Towed Demersal Fishing. PloS one 8:e83883.

Shephard, S., B. Beukers-Stewart, J. G. Hiddink, A. R. Brand, and M. J. Kaiser. 2010. Strengthening recruitment of exploited scallops *Pecten maximus* with ocean warming. Marine Biology 157:91–97.

Shephard, S., C. Goudey, A. Read, and M. Kaiser. 2009. Hydrodredge: Reducing the negative impacts of scallop dredging. Fisheries Research 95:206–209.

Stoner, A. W., and M. Ray-Culp. 2000. Evidence for Allee effects in an over-harvested marine gastropod: density-dependent mating and egg production. Marine Ecology Progress Series 202:297–302.

Szostek, C., A. Davies, and H. Hinz. 2013. Effects of elevated levels of suspended particulate matter and burial on juvenile king scallops *Pecten maximus*. Marine Ecology Progress Series 474:155–16

Tang, S. F. 1941. The breeding of the scallop (*Pecten maximus* (L.)) with a note on growth rate. Proceedings and Transactions of the Liverpool Biological Society 54:9–28.

Thurstan, R. H., and C. M. Roberts. 2010. Ecological meltdown in the Firth of Clyde, Scotland: two centuries of change in a coastal marine ecosystem. PloS one 5:e11767.

Vause, B., B. D. Beukers-Stewart, and A. R. Brand. 2006. Age composition and growth rates of queen scallops *Aequipecten opercularis* (L.) around the Isle of Man. Journal of Sea Research 25:310–312.

Vause, B. J., B. D. Beukers-Stewart, and A. Brand. 2007. Fluctuations and forecasts in the fishery for queen scallops (*Aequipecten opercularis*) around the Isle of Man. ICES Journal of Marine Science 64:1124–1135.

Veale, L., A. Hill, and A. Brand. 2000. An in situ study of predator aggregations on scallop (*Pecten maximus* (L.)) dredge discards using a static time-lapse camera system. Journal of experimental marine biology and ecology 255:111–129.

Veale, L., A. Hill, S. Hawkins, and A. Brand. 2001. Distribution and damage to the by-catch assemblages of the northern Irish Sea scallop dredge fisheries. Journal of the Marine Biological Association of the UK 81:85–96.

Warren, M., R. Gregory, B. Laurel, and P. Snelgrove. 2010. Increasing density of juvenile Atlantic (*Gadus morhua*) and Greenland cod (*G. ogac*) in association with spatial expansion and recovery of eelgrass (*Zostera marina*) in a coastal nursery habitat. Journal of Experimental Marine Biology and Ecology 394:154–160.

Watling, L., R. Findlay, L. Mayer, and D. Schick. 2001. Impact of a scallop drag on the sediment chemistry, microbiota, and faunal assemblages of a shallow subtidal marine benthic community. Journal of Sea Research 46:309–324.

Watling, L., and E. A. Norse. 1998. Disturbance of the seabed by mobile fishing gear: A comparison to forest clearcutting. Conservation Biology 12:1180–1197.

Watson, S.-A., P. C. Southgate, P. A. Tyler, and L. S. Peck. 2009. Early Larval Development of the Sydney Rock Oyster *Saccostrea glomerata* Under Near-Future Predictions of CO_2-Driven Ocean Acidification. Journal of Shellfish Research 28:431–437.

Wildish, D., G. Fader, P. Lawton, and A. MacDonald. 1998. The acoustic detection and characteristics of sublittoral bivalve reefs in the Bay of Fundy. Continental Shelf Research 18:105–113.

Zhou, S., A. D. M. Smith, A. E. Punt, A. J. Richardson, M. Gibbs, E. A. Fulton, S. Pascoe, C. Bulman, P. Bayliss, and K. Sainsbury. 2010. Ecosystem based fisheries management requires a change to the selective fishing philosophy. Proceedings of the National Academy of Sciences 107:9485–9489.